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Managing multiple ecosystem services

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Abstract

As human populations continue to grow, there is increasing demand to manage landscapes to increase the provisioning of multiple ecosystem services. However, this is challenging due to the negative (trade-off) and positive (synergistic) relationships that often exists among services. Understanding where and why these relationships occur should facilitate the implementation of better policies and strategies that can effectively manage multiple services simultaneously. However, our current understanding of what drives ecosystem service relationships, and the consequences for managing ecosystem services, remains limited. In this thesis, I address this by demonstrating the importance of understanding the drivers behind ecosystem service provision and the resulting trade-offs and synergies, and then apply this understanding to manage complex landscapes for multiple ecosystem services.

In ***chapter 1***, I discuss the conceptual theory behind ecosystem service relationships and how this information could be used to effectively manage multiple ecosystem services. I then outline the steps I undertake in this thesis to demonstrate this. I then conduct a literature review in ***chapter 2*** to determine how the drivers of ecosystem service relationships, and the mechanisms linking these drivers to ecosystem service outcomes, are currently being investigated in assessments of ecosystem service synergies and trade-offs. I find that the majority of assessments of ecosystem service relationships do not explicitly identifying the drivers of these relationships. This is strongly related to the methods used to identify the trade-offs and synergies, with the less commonly used process-based approaches better equipped to explicitly identify the drivers underpinning ecosystem service relationships.

I then develop models to identify the drivers underpinning the provisioning of multiple cultural ecosystem services in urban public greenspace, using the urban parks network of Brisbane, Australia, as a case study in ***chapter 3***. Using data derived from a social survey and remotely sensed data, I demonstrate that the use of public parks for four cultural services (opportunities for physical exercise, nature interactions, relaxation and social interactions) are associated with spatial, environmental and facility characteristics of urban parks. However, physical exercise and social interactions are also driven by the socio-demographic characteristics of the people visiting the parks to receive these services. These results suggest that by introducing management actions that target specific variables within urban parks it may be possible to facilitate the provision of multiple cultural ecosystem services simultaneously.

I then build upon the findings of ***chapter 3*** to assess the implications of ecosystem service trade-offs and synergies on the management of multiple ecosystem services across the Brisbane parks network in ***chapters 4*** and ***5***. In ***chapter 4***, I conduct a scenario analysis to identify the trade-offs and synergies among the cultural ecosystem services identified in ***chapter 3*** and carbon storage, and how these relationships vary under different revegetation management strategies commonly implemented in urban parks. I find that the relationships among the ecosystem services depend on the type of revegetation management strategy considered. This indicates that careful consideration of relationships among services could ensure the implementation of strategies that minimise trade-offs among services. In ***chapter 5*** I use this underlying model to identify the optimal spatial allocation of multiple management actions to achieve targeted increases in the provision of carbon storage and the same set of cultural services across the Brisbane park network. I also consider the consequences of considering social equity in ecosystem service access and management and compare socially equitable and inequitable optimal solutions. I find that implementing strategies that consist of multiple management actions achieve greater increases in multiple ecosystem services. Furthermore, I find that accounting for social equity restricts the extent to which ecosystem service can be increased to, while also increasing management costs.

Finally, in ***chapter 6*** I synthesise the main findings of the previous chapters, and discuss the contributions of this thesis to the literature and future research directions. Currently, few studies explicitly identify the drivers of ecosystem service relationships, but this thesis demonstrates that considering drivers is vital to managing multiple ecosystem services effectively. Although this can be challenging, explicitly incorporating these drivers into assessments of ecosystem service relationships can ensure more effective management of multiple ecosystem services across landscapes. Ideally, to improve the management of multiple ecosystem services simultaneously, future research should focus on working towards more causally-explicit approaches to identify ecosystem service relationships, and on incorporating social equity into ecosystem service management strategies.

Declaration by author

This thesis is composed of my original work, and contains no material previously published or written by another person except where due reference has been made in the text. I have clearly stated the contribution by others to jointly-authored works that I have included in my thesis.

I have clearly stated the contribution of others to my thesis as a whole, including statistical assistance, survey design, data analysis, significant technical procedures, professional editorial advice, financial support and any other original research work used or reported in my thesis. The content of my thesis is the result of work I have carried out since the commencement of my higher degree by research candidature and does not include a substantial part of work that has been submitted to qualify for the award of any other degree or diploma in any university or other tertiary institution. I have clearly stated which parts of my thesis, if any, have been submitted to qualify for another award.

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Publications included in this thesis

Brown, G., Rhodes, J. and **Dade, M**. 2018. An evaluation of participatory mapping methods to assess urban park benefits. *Landscape and Urban Planning* 178: 18-31. https://doi.org/10.1016/j.landurbplan.2018.05.018
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Dade, M.C., Mitchell, M.G.E., McAlpine, C.A. and Rhodes, J.R. Assessing ecosystem service trade-offs and synergies: the need for a more mechanistic approach. *Ambio*. https://doi.org/10.1007/s13280-018-1127-7

Conference abstracts:

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Dade, M.C., Brown, G., Mitchell, M.G.E. and Rhodes. J.R. 2017. Identifying trade-offs among recreational ecosystem services in urban greenspaces. *EcoTas 2017*, Pokolbin, New South Wales, Australia.

Dade, M.C., Brown, G., Mitchell, M.G.E. and Rhodes. J.R. 2017. Identifying trade-offs among recreational ecosystem services in urban greenspaces. *International Congress for Conservation Biology*, Cartagena, Colombia.

Dade, M.C., Mitchell, M.G.E., McAlpine, C. and Rhodes, J.R. 2016. Moving towards a causative approach to characterising ecosystem service trade-offs and synergies. *EcoSummit*, Montpellier, France.

Dade, M.C., Mitchell, M.G.E., McAlpine, C. and Rhodes, J.R. 2016. Defining the relationship; Moving towards a causative approach to identify and quantify ecosystem service trade-offs and synergies. *Society for Conservation Biology 4th Oceania Congress*, Brisbane, Australia.

Contributions by others to the thesis

This thesis consists of three manuscripts that are intended for submission for publication and one manuscript that has been published, with myself as the lead author, and one manuscript in Appendix A that has been published, with myself as a co-author. Chapters 2 – 5 and all appendices are written using plural first-person pronoun "we"/"our", to reflect the contributions from others. In chapters 1 and 6, I use the singular first-person pronoun "I"/"my" as these were written entirely by me (with editorial input from my supervisors).

Chapter 1

This chapter was written by myself, with editorial input from Jonathan Rhodes, Matthew Mitchell and Clive McAlpine.

Chapter 2

This chapter has been published in *Ambio*, and further information on the publication is detailed in the “Publications included in this thesis” section. The idea for the chapter was conceptualised and designed by Jonathan Rhodes, Matthew Mitchell, Clive McAlpine and myself. The literature review and analysis of data was conducted by myself, with advice from Jonathan Rhodes. The chapter was written by myself, with editorial input from Jonathan Rhodes, Matthew Mitchell and Clive McAlpine.

Chapter 3

This chapter is being prepared for submission to *Ecosystem Services*. The idea for the chapter was conceptualised by Jonathan Rhodes, Matthew Mitchell and myself. The social survey was conducted by Greg Brown, Jonathan Rhodes and myself. Data analyses were conducted by myself, with advice from Jonathan Rhodes and Matthew Mitchell. The chapter was written by myself, with editorial input from Jonathan Rhodes, Matthew Mitchell and Greg Brown.

Chapter 4

This chapter is being prepared for submission to *Ecosystem Services*. The idea for the chapter was conceptualised by Jonathan Rhodes, Matthew Mitchell and myself. The modelling of ecosystem services was conducted by myself, with input from Matthew Mitchell. The scenario analysis was conducted by myself, with advice from Matthew Mitchell and Jonathan Rhodes. The chapter was written by myself, with editorial input from Jonathan Rhodes and Matthew Mitchell.

Chapter 5

This chapter is being prepared for submission to *Nature Sustainability*. The idea for the chapter was conceptualised by Jonathan Rhodes and myself. The optimisation analysis of ecosystem services was conducted by Jeffrey Hanson and myself, with advice from Jonathan Rhodes. All other analyses were conducted by myself, with advice from Jonathan Rhodes. This chapter was written by myself, with editorial input from Jonathan Rhodes and Matthew Mitchell.

Chapter 6

This chapter was written by myself, with editorial input from Jonathan Rhodes and Matthew Mitchell.

Appendix A

This chapter has been published in *Landscape and Urban Planning*, and further information on the publication is detailed in the “Publications included in this thesis” section. The idea for the chapter was conceptualised by Greg Brown. Jonathan Rhodes and myself. The data was collected by Greg Brown, Jonathan Rhodes and myself. The analysis of the data was conducted by Greg Brown. The chapter was written by Greg Brown, with editorial input from Jonathan Rhodes and myself.

Statement of parts of the thesis submitted to qualify for the award of another degree

None.

Research Involving Human or Animal Subjects

This thesis includes research involving human subjects. This research consists of a survey undertaken in Chapter 3 which was conducted with human research ethics approval provided to an ARC Discovery Project (DP 130100218), of which this thesis is a part of. Ethical clearance was provided by the University of Queensland’s Human Ethics Unit, and approved by the Chairperson of the Ethics Committee, Professor Emerita Gina Geffen (approval number 2016001148). A copy of the ethics approval letter is provided in Appendix B.

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Keywords

Ecosystem services, trade-offs, synergies, urban planning, spatial planning, carbon storage, cultural ecosystem services, environmental policy and planning.

Australian and New Zealand Standard Research Classifications (ANZSRC)

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ANZSRC code: 050205, Environmental Management (30%)
ANZSRC code: 050104, Landscape Ecology (20%)

Fields of Research (FoR) Classification

FoR code: 1205, Urban and Regional Planning (50%)
FoR code: 0502, Environmental Science and Management (30%)
FoR code: 0501, Ecological Applications (20%)

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CHAPTER 1
INTRODUCTION

Ecosystem services are the mental and physical benefits that humans derive from ecosystems (Daily et al. 1997). As human populations continue to grow, demand for ecosystem services is also increasing (United Nations 2015). Therefore, strategies are required to sustainably manage the multiple ecosystem services realized for humans in socio-ecological systems. However, this is challenging due to the complex relationships that exist among ecosystem services (Rodríguez et al. 2006). Taking an action that alters the provisioning of one ecosystem service can lead to, positive or negative, changes in the provisioning of other ecosystem services. Currently, our development of effective management strategies is hindered by limited knowledge of when and how different ecosystem services respond to drivers of change and whether this leads to synergies or trade-offs among services (Kremen and Ostfeld 2005). Without this knowledge, we run the risk of introducing unsustainable management strategies that lead to unexpected declines in ecosystem service provisioning, and consequent decreases in human wellbeing (Gaston et al. 2013; Lindenmayer et al. 2012).

To identify strategies that can sustainably achieve the provisioning of multiple ecosystem services across landscapes requires undertaking two main steps. First, it is necessary to understand how management actions influence the provisioning of multiple ecosystem services and potentially lead to positive and negative relationships occurring between the services. Secondly, given these identified linkages between management actions and ecosystem service relationships, it is necessary to identify when and where to implement these actions that ensure the provisioning of multiple ecosystem service increases. For example, deforestation can increase areas for cattle grazing and therefore drive an increase in meat production, an ecosystem service, but also drives a decrease in carbon storage due to the loss of woody biomass (Coomes et al. 2008). Due to the trade-off between the ecosystem services under deforestation, this management action may be unable to manage the provisioning of both ecosystem services simultaneously. To achieve the above steps, methods are required that can quantify the linkages between multiple ecosystem services, and the drivers and mechanisms underpinning these relationships (Bennett et al., 2009). Therefore to sustainably manage multiple ecosystem services requires understanding these interactions between ecosystem services and management strategies.

In the following sections I outline the background and motivation for this thesis followed by a description of the structure of the thesis. This background section is divided into five sub-sections to describe our current knowledge around identifying and managing multiple ecosystem services, and the relationships that exist among them. The first section, ‘Ecosystem services’ gives a general overview of ecosystem services. The second section, ‘Trade-offs, synergies, and the drivers underpinning them’ discusses the relationships that exist between ecosystem services and the factors that can influence these relationships. The third section, ‘Identifying trade-offs and synergies’ discusses the implications of understanding when and where trade-offs and synergies occur. The fourth section, ‘Managing ecosystem service relationships in urban landscapes’ discusses the nature of ecosystem service trade-offs and synergies in complex landscapes, using urban landscapes as an example, and the implications this has for managing multiple services simultaneously. The fifth section, ‘Spatial conservation prioritisation for managing ecosystem services’ discusses how spatial conservation optimisation tools can be applied to effectively manage multiple ecosystem services across landscapes. The final section identifies the key research gaps and outlines the objectives and structure of this thesis.

1.1 ECOSYSTEM SERVICES

Ecosystem services are generally categorised into three groups: provisioning, regulating, and cultural (Costanza et al. 2017; MA 2005). Provisioning services are the material goods and resources provided by ecosystems, such as food, timber, medicinal resources and fresh drinking water (de Groot et al. 2002). Regulating services are provided by processes that regulate ecosystems and benefit human wellbeing, such as temperature regulation, climate regulation, pollination and pest regulation (de Groot et al. 2002). Finally, cultural services refer to the non-material benefits people obtain from ecosystems, such as aesthetic appreciation, different forms of recreation, and cultural identity (Daniel et al. 2012).

All these ecosystem services are generated by the living and non-living components of ecosystems, also known as natural capital (Guerry et al. 2015) (see Table 1.1 for a glossary of definitions). For example, forests are a form of natural capital that supply carbon storage, and also opportunities for exercise (Maseyk et al. 2017). A demonstrated in Figure 1.1, for people to benefit from an ecosystem service (referred to as ecosystem service provisioning), the absolute potential amount of a service that an ecosystem can provide (referred to as the supply) must connect with the people who desire or require this service (referred to as demand) (Mitchell et al. 2015; Villamagna et al.

2013). This movement between supply and demand is referred to as ecosystem service flow. Flow can result from the movement of organisms, such as the movement of pollinators achieves pollination (Kremen et al. 2007). It can result from the movement of people, for example, for aesthetic value to be realized from a landscape within a national park, people must travel to the location of the view (Martinez-Harms et al. 2018). Flow can also result from the use of technology that can connect people to ecosystem services (e.g., water pipes (Liu et al. 2016)). Thus, natural capital, supply, flow and demand must all be present for a component of an ecosystem to become an ecosystem service.

Table 1.1 Glossary of common terms used to conceptually describe the different components of ecosystem service delivery. Adapted from Mitchell et al. (2015).

Term	**Definition**
Ecosystem service	The physical and mental benefits people receive from ecosystems.
Natural capital	The stock of natural ecosystems, including all of their biological and physical features that generate ecosystem services.
Ecosystem service supply	The full potential of ecological functions or biophysical elements in an ecosystem to provide a given ecosystem service, without consideration of whether humans recognize, use, or value it.
Ecosystem service demand	The amount of an ecosystem service desired or required by people. Demand is influenced by human needs, values, institutions, built capital, and technology.
Ecosystem service flow	The movement of an ecosystem service to people, or vice versa. Ecosystem service flow depends on both the supply of and demand for a service as well as the movement of organisms, matter, and people.
Ecosystem service provisioning	When an ecosystem service is provided to a person. For provisioning to occur, ecosystem service supply and demand must connect.
Ecosystem service benefit	The ways in which ecosystems improve human well-being via the provision of ecosystem services. This includes materials essential for life and contributions to health, security, social relations, and freedom of choice and action.

Ecosystems can provide multiple ecosystem services, depending on the presence of natural capital, and ecosystem service supply, flow and demand. Often multiple services are provided by the same natural capital stock, or have multiple different beneficiaries (Villamagna et al. 2013). For example, a forest, a natural capital stock, could provide a source of timber to one person, and a source of recreation to another (Ninan and Inoue 2013). Any changes in the components of an ecosystem can lead to changes in the natural capital and the supply, flow and demand which can increase or decrease how much of multiple ecosystem services can be provided.

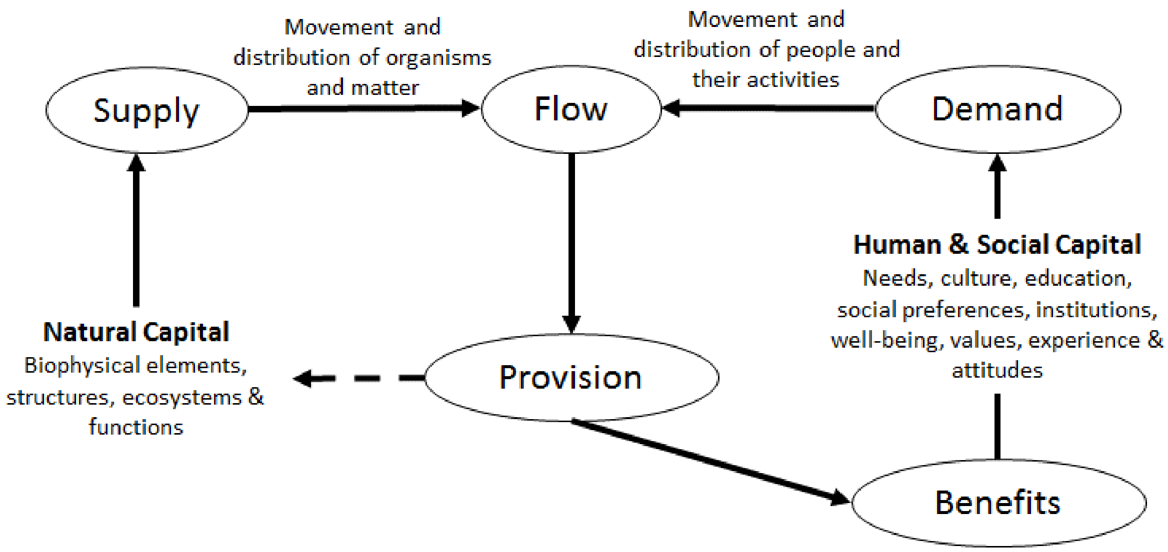

Figure 1.1 A conceptual diagram outlining the components of ecosystem service delivery. The dashed line represents an indirect effect. Adapted from Mitchell et al. (2015).

1.2 TRADE-OFFS, SYNERGIES, AND THE DRIVERS UNDERPINNING THEM

Ecosystem services can interact with one another (Rodríguez et al. 2006). This means that a change in the provisioning of one ecosystem service can often also result in an increase or decrease in the provisioning of multiple other services. These relationships arise in response to an exogenous or endogenous change to the system, known as a driver, that leads to changes in ecosystem components or processes, or the pathways by which ecosystem services are delivered (e.g., the movement of species, or the movement of people), resulting in changes in ecosystem service provisioning (Bennett et al. 2009; Duncan et al. 2015; Mitchell et al. 2015). These drivers can include, among others, policy instruments, natural environmental variation, human behaviour and technological advances. These ecosystem service relationships can be in the form of trade-offs, where the provisioning of one service increases as another decreases, synergies, where the provisioning of two services increase or decrease simultaneously, or there can be no relationship, where there is no link between the provisioning of two services (Rodríguez et al. 2006). For example, reforestation (a driver) can increase carbon storage but could decrease food provisioning, as the area for cropland decreases, leading to a trade-off (McKinley et al. 2011). On the other hand, restoring riparian vegetation (a driver) will increase both flood mitigation and water quality regulation, leading to a synergy (Richardson et al. 2007).

The relationships between ecosystem services – whether they are trade-offs or synergies – can vary between the ecosystem services and contexts (Duncan et al. 2015; Lee and Lautenbach 2016; Seppelt et al. 2013). Therefore, two different drivers could potentially generate two different types of relationships between the same two ecosystem services. For example, increasing agroforestry may act as a driver promoting a synergy between biodiversity and carbon storage (Jose et al. 2009), while increasing mono-culture plantations may drive a trade-off between the same ecosystem services (Potvin et al. 2011).

Though drivers are crucial to the existence of trade-offs and synergies, there is little information on how often assessments on ecosystem service relationships consider them. Lee and Lautenbach (2016) reviewed assessments on ecosystem service relationships, and though they identified a diversity in the relationships occurring among the same ecosystem services, there was no assessment of whether these studies explicitly identified the drivers causing these relationships. A large number of ecosystem service assessments have also focused on the social and ecological factors that form 'bundles' of ecosystem services to co-occur spatially or temporally (Raudsepp-Hearne et al. 2010; Spake et al. 2017). However, these studies only capture the spatial and temporal pattern in ecosystem services, not the ecological drivers underlying the provisioning of each ecosystem service and the emergent relationships among services (Spake et al. 2017). Furthermore, there is bias in our knowledge of the processes underpinning certain ecosystem services. For example, while a large amount of research focuses on the ecological processes that supply provisioning and regulatory ecosystem services, there is very little information available on the processes that underpin the provisioning of cultural ecosystem services (Milcu et al. 2013).

1.3 IDENTIFYING TRADE-OFFS AND SYNERGIES

Understanding when and where trade-offs and synergies occur, and the drivers underpinning them, can help achieve more effective management of multiple ecosystem services. Identifying a trade-off or synergy between two ecosystem services can allow us to predict which ecosystem services are likely to decline or increase when certain management actions are introduced, and the impacts that managing one ecosystem service will have on the provisioning of multiple other services (Briner et al. 2013). For example, using fertiliser in agricultural systems can drive an increase in food production but a decrease in water quality due to runoff (Smith and Siciliano 2015). Therefore, by understanding that a trade-off occurs and what drives it, the policy makers and regulators are able to make an informed decision whether to continue or restrict fertiliser use, and the consequences this will have on multiple ecosystem services. This knowledge can prevent unexpected declines in the provisioning of ecosystem services important to human wellbeing when management is changed.

Importantly, in a management context, identifying trade-offs and synergies, and the drivers underpinning them, can allow us to predict which ecosystem services are likely to increase or decrease under different management scenarios. Management actions can act as drivers of ecosystem service trade-offs and synergies as they alter ecosystems that play a role in ecosystem service provisioning, and alter demand for ecosystem services through incentives (Briner et al. 2013). Failing to identify the role that management actions play in driving the relationships among ecosystem services can make it difficult to determine the consequences management actions will have on the provisioning of multiple ecosystem services, and can lead to poor management decisions. This could be both economically and environmentally costly, potentially leading to unexpected declines in ecosystem services that are vital to human wellbeing (Lindenmayer et al. 2012). Such perverse outcomes can be avoided by explicitly considering drivers into assessments of ecosystem service trade-offs and synergies (Bennett et al. 2009; Briner et al. 2013; Rodríguez et al. 2006). Identifying trade-offs and synergies among a group of ecosystem services under alternate management strategies prior to implementation could ensure effective strategies are introduced that minimise trade-offs, and declines in ecosystem service provisioning.

There are a huge variety of methods available to identify ecosystem service trade-offs and synergies, ranging from simple correlation coefficients to process-driven models (Mouchet et al. 2014). Though previous studies have synthesised these methods (Lee and Lautenbach 2016; Mouchet et al. 2014), it remains unclear how often drivers are being considered in assessments of ecosystem service trade-offs and synergies. Furthermore, there is limited information on the methods that are capable of explicitly incorporating drivers into assessments of trade-offs and synergies. A clearer understanding of these methods will allow us to better choose which methods to employ that can determine the outcome of different policies and landscape processes on ecosystem service relationships, enabling more efficient management actions and policies to be implemented to manage the provisioning of these services.

1.4 MANAGING ECOSYSTEM SERVICE RELATIONSHIPS IN URBAN LANDSCAPES

Managing multiple ecosystem services in human-dominated and multifunctional landscapes is particularly difficult due to a wide array of potential drivers present and our often limited knowledge on the relationships occurring among the ecosystem services in these landscapes (Bennett et al. 2015; Birkhofer et al. 2015). This is a particular challenge for urban areas which are increasing rapidly to accommodate a growing population (United Nations 2015). It is important that the provisioning of multiple ecosystem services are managed effectively within these landscapes to meet the growing demand for a range of services (Kremer et al. 2016). To achieve this requires an in depth understanding of the drivers underpinning ecosystem service trade-offs and synergies.

Urban landscapes are dynamic, providing a multitude of ecosystem services that are crucial to the wellbeing of urban residents, with an ever-changing list of drivers present that can result in different relationships occurring among the services (Ramalho and Hobbs 2012). For example, urban landscapes can be managed under a wide array of policies and can contain a wide variety of other processes that act as drivers, affecting the trade-offs and synergies that occur between urban ecosystem services. These can include urban development policies, public greenspace management, vegetation restoration policies, socio-economic status of residents, landscape structure, elevation and transport policies (Grimm et al. 2008; Li et al. 2010; Stott et al. 2015; Tratalos et al. 2007). For example, an urban expansion policy (a driver) could reduce the area available for vegetation, decreasing both carbon storage and pollination, and therefore create a synergy between these services (Dobbs et al. 2014a; Tratalos et al. 2007). On the other hand, environmental restoration policies can also act as drivers of a synergy between these same ecosystem services, but to a different degree (Standish et al. 2013). Therefore, there are a variety of drivers present in urban landscapes, affecting ecosystem service relationships.

For an ecosystem service to provide a benefit to a person, the person must have access to the supply of the service (Villamagna et al. 2013). In urban landscapes this can be difficult, as there are multiple land uses, often a lack of access and socio-economic factors that can influence who has access to ecosystem services, and who does not. For example, Jenerette et al. (2011) found that, within urban areas, people living in high socio-economic neighbourhoods have greater access to air temperature regulation due to the high proportion of tree cover in these neighbourhoods. Therefore, to implement effective management of multiple ecosystem services across urban landscapes, it is important that management actions are implemented in such a way to ensure that benefits from ecosystem service provisioning are socially equitable. However, research into equitable management actions to protect conservation features have found that the conservation targets achievable decrease when social equity is accounted for (Halpern et al. 2013). Though previous studies have discussed the need for increased consideration of social equity in managing ecosystem services (Jennings et al., 2016; Martinez-Harms et al. 2018), no studies have assessed the impact of considering social equity when spatially allocating management actions to improve ecosystem services in complex landscapes, such as urban landscapes.

1.5 SPATIAL CONSERVATION PRIORITISATION FOR MANAGING ECOSYSTEM SERVICES

Ensuring that management actions increase multiple ecosystem services to target levels, and that these increases are distributed equitably across the landscape, is a complex spatial optimisation problem (Jennings et al. 2017; Schröter and Remme 2016; Snäll et al. 2016). It requires understanding what targets for the provisioning of multiple ecosystem services need to be achieved, what management actions are available to be implemented to reach these targets, and how much do each of these actions change the provisioning of each ecosystem service across the landscape (Schröter and Remme 2016). A number of studies have previously focused on using spatial conservation planning tools, such as Marxan and Zonation to spatially optimise ecosystem service provision (see Luck et al. (2012)). For example, Chan et al. (2006) identified the spatial priority areas for ecosystem service provisioning across a multifunctional region. Law et al. (2016) identified the effect of different land use strategies on achieving ecosystem service targets within a multi-use region of Borneo, Indonesia. However, our knowledge on spatially optimising the provisioning of multiple ecosystem services is still limited (Snäll et al. 2016). There is often a wide variety of management actions available to manage ecosystem services, however, no study has used spatial optimisation tools to identify where to allocate multiple different management actions simultaneously across a landscape to achieve multiple ecosystem service targets. This information will allow decision makers to determine what management actions to implement, and where, to achieve target increases in multiple ecosystem services (Schröter and Remme 2016; Snäll et al. 2016). Furthermore, by also considering the costs of the different management actions and the spatial distribution of the increases in ecosystem service provisioning, it is possible to identify where to allocate management actions to achieve targets in multiple ecosystem services, at minimum cost and also ensure social equity across the landscape. This information will ensure management actions create sustainable landscapes, where environmental and economic benefits are achieved alongside social equity (Wu 2013).

1.6 THESIS OBJECTIVES AND SIGNIFICANCE

This thesis explicitly evaluates the importance of understanding the drivers behind ecosystem services, and the resulting trade-offs and synergies, and applies this understanding to managing complex landscapes for multiple ecosystem services. Currently, no analysis exists that evaluates how often drivers are considered in assessments of ecosystem service trade-offs and synergies, and few of the drivers underpinning ecosystem service relationships, particularly those involving cultural services, have been identified. Furthermore, the impacts of different management actions on ecosystem service relationships within urban landscapes has not been evaluated, and no studies have assessed how to spatially allocate management actions across multifunctional landscapes to equitably provide multiple ecosystem services. To address these knowledge gaps, this thesis addresses four main objectives (Figure 1.2):

1. Identify how often drivers and mechanisms linking drivers to ecosystem services are considered in assessments of the synergies and trade-offs among ecosystem services and provide recommendations for improving these assessments.
2. Understand the drivers underpinning the provisioning of multiple cultural ecosystem service within urban landscapes.
3. Determine how the trade-offs and synergies vary under different urban park management actions.
4. Apply an understanding of the links between drivers and ecosystem services to maximise provisioning of multiple ecosystem services and equity objectives.

To achieve objective 1, I completed a systematic literature review of how drivers have been considered in assessments of ecosystem service trade-offs and synergies (*Chapter 2*). Furthermore, this determines whether methods used to identify trade-offs and synergies are related to how drivers are considered and which drivers are most commonly being assessed.

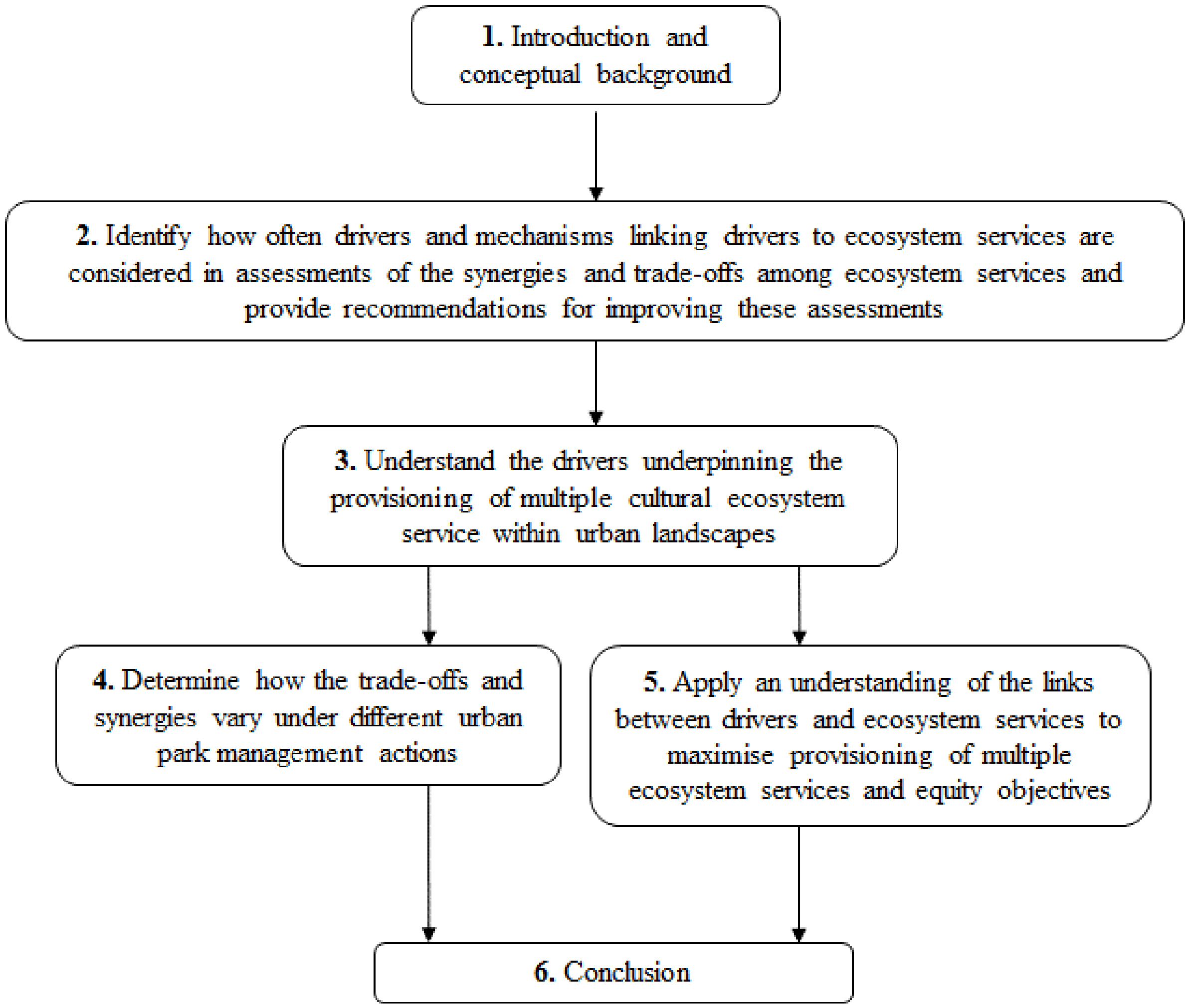

Figure 1.2 Flowchart of the thesis structure. Solid boxes represent chapters, arrows indicate the linkages between chapters and numbers represent the thesis chapters.

This review is followed by three data analysis chapters focusing on an urban landscape, using Brisbane, Australia, as a case study (Figure 1.2). To effectively manage trade-offs and synergies among ecosystem services, it is necessary to first determine the drivers underpinning these relationships (Duncan et al. 2015). However, though the drivers that provide many provisioning and regulatory services in urban areas are well documented, the processes providing cultural ecosystem services remain relatively unknown (Kremer et al. 2016; Milcu et al. 2013). *Chapter 3* (Objective 2) addresses this by identifying the socio-demographic and urban park characteristics that influence which cultural ecosystem services are provided by urban parks. The next two chapters focus on the effective management of multiple ecosystem services, while considering the drivers of relationships that exist among them. *Chapter 4* (Objective 3) uses the models developed in *Chapter 3* to identify the trade-offs and synergies that exist among a group of urban ecosystem services, and how these relationships change as different management actions are introduced to urban parks. In the next chapter (*Chapter 5*, Objective 4) I further build upon *Chapter 3* to identify the optimal spatial allocation of management actions that achieves targeted in increases in multiple ecosystem services, while accounting for social equity in management and access to these ecosystem services.

The management of multiple ecosystem service within landscapes requires understanding the drivers underpinning ecosystem service relationships. However, identifying the linkages between drivers and ecosystem services can be a challenging task, particularly in complex landscapes, such as urban landscapes, where ecosystems are constantly being altered. In this thesis, I address these gaps by developing and evaluating methods and approaches to manage multiple ecosystem services across multifunctional landscapes.

CHAPTER 2

ASSESSING ECOSYSTEM SERVICE TRADE-OFFS AND SYNERGIES: THE NEED FOR A MORE MECHANISTIC APPROACH

This chapter is reproduced from the following paper, with some alterations to formatting and structure:

> Dade, M.C., Mitchell, M.G.E., McAlpine, C.A. and Rhodes, J.R. Assessing ecosystem service trade-offs and synergies: the need for a more mechanistic approach. *Ambio*. https://doi.org/10.1007/s13280-018-1127-7

The idea for the manuscript was conceptualised and designed by Jonathan Rhodes, Matthew Mitchell, Clive McAlpine and myself. The literature review and analysis of data was conducted by myself, with advice from Jonathan Rhodes. The manuscript was written by myself, with editorial input from Jonathan Rhodes, Matthew Mitchell and Clive McAlpine.

2.1 ABSTRACT

Positive (synergistic) and negative (trade-off) relationships among ecosystem services are influenced by drivers of change, such as policy interventions and environmental variability, and the mechanisms that link these drivers to ecosystem service outcomes. Failure to account for these drivers and mechanisms can result in poorly informed management decisions and reduced ecosystem service provision. Here, I review the literature to determine the extent to which drivers and mechanisms are considered in assessments of ecosystem service relationships. I show that only 13% of assessments explicitly identify the drivers and mechanisms that lead to ecosystem service relationships. While the proportion of assessments considering drivers has increased over time, most of these studies only implicitly consider drivers and the mechanisms linking drivers to ecosystem services. To ensure effective management of ecosystem services, I recommend greater use of causal inference and process-based models for advancing our understanding of ecosystem service trade-offs and synergies.

2.2 INTRODUCTION

One of the goals of many environmental policy initiatives is improved human well-being through the provision of ecosystem service benefits from natural and human modified ecosystems (Guerry et al. 2015; Kandziora et al. 2013; MA, 2005; Raudsepp-Hearne et al., 2010; Tallis et al., 2008). Initiatives such as the United Nation’s Sustainable Development Goals (SDGs) and the Reduced Emission from Deforestation and Environmental Degradation (REDD+) focus on managing multiple ecosystem services (Alexander et al. 2011; Griggs et al. 2013). Considering multiple services complicates policy decisions because complex relationships exist among ecosystem services that can lead to simultaneous positive and negative changes in the provision of different ecosystem services in response to a policy change (Bennett et al. 2009; Howe et al. 2014). Understanding these relationships among ecosystem services to inform policy is therefore important, but it requires a consideration of the specific drivers of change (such as policy interventions) and the mechanisms that link drivers to ecosystem service outcomes across multiple services. Theoretical and conceptual models have been developed to help us understand the mechanisms that determine the relationships between ecosystem services (Bennett et al. 2009; de Groot et al. 2010; Rounsevell et al. 2010; Villamagna et al. 2013). However, the extent to which these mechanisms are considered in empirical assessments of ecosystem service relationships is unclear.

Ecosystem service relationships can occur as trade-offs, where the provisioning of one service increases as another decreases, or as synergies, where the provisioning of two services increase or decrease simultaneously (Rodríguez et al. 2006). These relationships arise in response to exogenous or endogenous changes to the system, referred to as drivers (Bennett et al. 2009) that can be related to human interventions and natural variability, including policy instruments, climate change, and technological advances. For example, Schröter et al. (2005) determined that climate change drives a trade-off between two ecosystem services, carbon storage and food production in Europe, as it increases the suitable area for forests while decreasing the area suitable for arable land. The biotic and abiotic mechanisms that link drivers to the provision of ecosystem services, are also crucial to the presence of trade-offs or synergies between ecosystem services (de Groot et al. 2002; Potschin and Haines-Young 2011). For example, increasing temperatures (a driver) in boreal forests resulting from global climate change have been found to decrease the rate of soil nutrient cycling. Since this rate is a mechanism that affects two final ecosystem services, below ground carbon storage and maintenance of soil fertility, increasing temperatures create a negative synergy between these two services (Allison and Treseder 2008). Thus, identifying drivers and the mechanisms linking drivers to ecosystem services is key to understanding whether trade-offs or synergies between services are likely to occur.

Bennett et al. (2009) developed a framework for understanding how drivers can influence ecosystem service provision through different mechanistic pathways and hence the relationships among services. They outlined four main mechansitic pathways by which drivers can affect ecosystem service relationships. First, a driver can directly affect the supply of one ecosystem service, with no effect on another ecosystem service. Second, a driver can affect a single ecosystem service that has a unidirectional (one way) or bidirectional (two way) interaction with another ecosystem service. Third, a driver can directly affect two ecosystem services that do not interact with each other. Fourth, a driver can directly affect two ecosystem services that also have either a unidirectional or bidirectional interaction between them.

An important insight from Bennett et al. (2009) is that trade-offs and synergies between ecosystem services can vary depending on the drivers and mechanistic pathways that link drivers to ecosystem services (Figure 2.1). For example, a policy for reforesting abandoned cropland, where there is no competition between forest and cropland, will result in an increase in carbon sequestration, but with no direct effect on food production (Rey Benayas et al. 2007). Consequently, this represents the first pathway of the Bennett et al. (2009) framework (Fig. 2.1a) with no trade-off or synergy occurirng between these services. Contrastingly, a forest restoration policy such as the Grain to Green program in China (Liu et al. 2008), may incentivise reforestation and lead to increased carbon sequestration, but could also lead to decreased food production due to competition for land as cropland is replaced by forest. This interaction between the two services could result in a trade-off between carbon sequestration and food production (Figure 2.1(b)). In comparison, a policy promoting the restoration of riparian vegetation within agricultural landscapes, where riparian zones are often unsuitable for agriculture and there is little competition between these two land uses, could lead to both increased carbon sequestration as tree cover increases, and increased crop production since riparian vegetation can improve soil retention an improve crop production (Stutter et al. 2012). Therefore, a synergy results between the services despite no direct interaction between the two (Figure 2.1(c)). Alternatively, a policy that incentivises urban expansion could negatively affect the area of both forests and croplands and result in a negative synergy (Figure 2.1(d)) through the fourth mechanistic pathway (Lawler et al. 2014). However, if there is a subsequent expansion of crops at the expense of forest to meet food demand, and a strong negative interaction between the two services is created, then a trade-off between the two services is also possible.

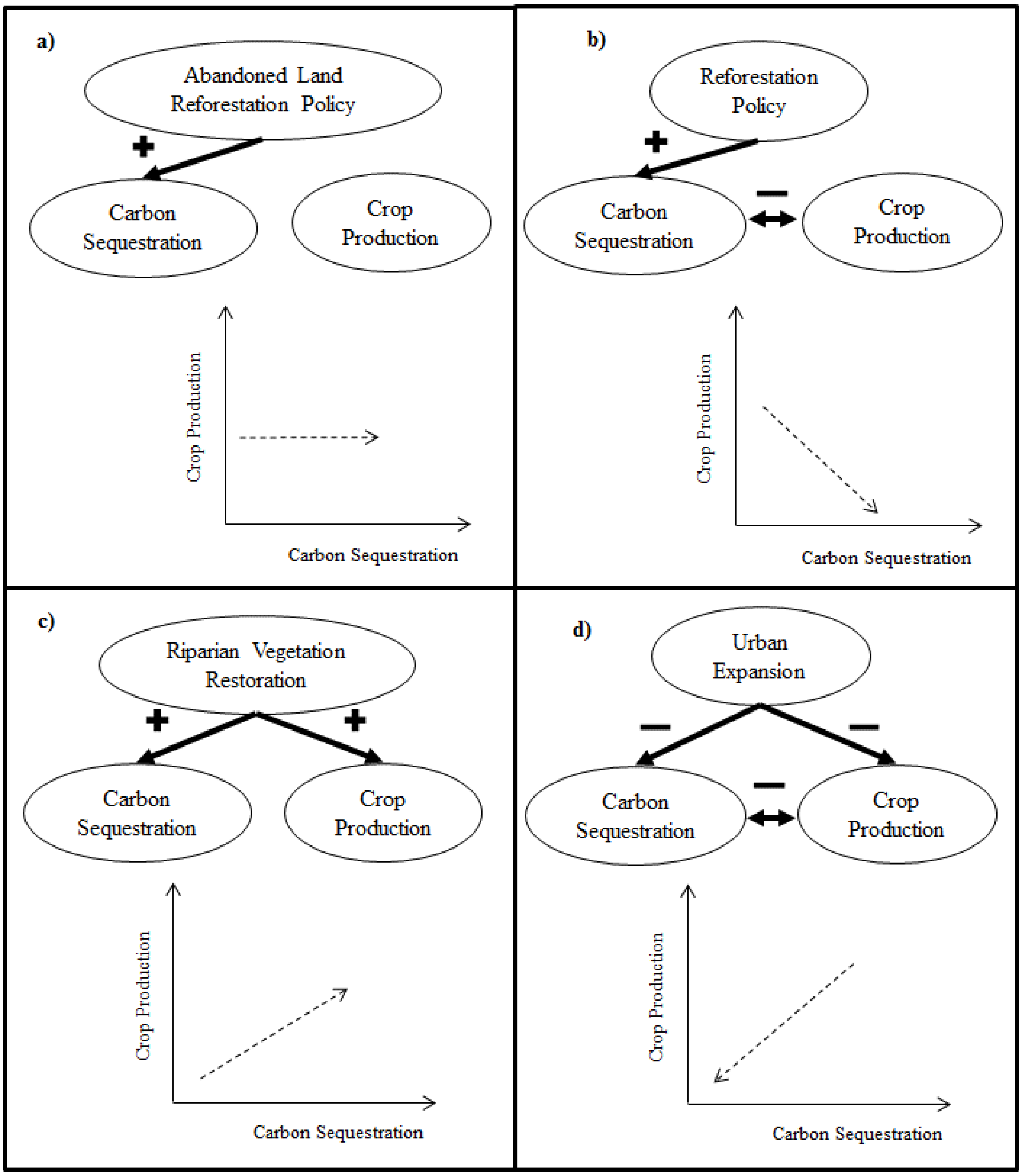

Figure 2.1 Schematic demonstrating how the mechanistic pathways in which drivers affect ecosystem services can affect the relationships between ecosystem services. In a), the reforestation of abandoned agricultural land (the driver) increases forested area, and consequently carbon sequestration. However, since this land is no longer used for crop production, this reforestation has no effect on food production. Therefore, there is no trade-off or synergy between the two ecosystem services. In b), a restoration policy (the driver) positively affects the forested area. However, because cropland and forest compete for land, forest area increases at the expense of cropland. This leads to a trade-off between the two ecosystem services. In c), management actions to restore degraded riparian vegetation will increase carbon sequestration due to increased tree cover, and increase crop production as it increases soil fertility, creating a synergy. As riparian zones are often unsuitable for agriculture, there is no competition between the two ecosystem services under this management action. In d), urban expansion (the driver) negatively affects the area available for both cropland and forest. Cropland and forest also negatively interact with one another as they compete for land. However, since the driver simultaneously decreases the area available for cropland and forests, this leads to a negative synergy between carbon sequestration and food production.

Given that different drivers and mechanistic pathways lead to very different synergistic or trade-off outcomes, failing to incorporate a mechanistic understanding into the assessment of ecosystem service trade-offs and synergies is likely to lead to misleading inferences. In the context of policy drivers, this can lead to poor decisions and unexpected declines in ecosystem services (Lindenmayer et al. 2012). There have been a number of reviews on the assessment of trade-offs and synergies (Howe et al. 2014; Lee and Lautenbach 2016; Mouchet et al. 2014; Spake et al. 2017), but none have quantified the extent to which drivers and mechanisms are considered in these assessments.

In this chapter, I address this limitation by systematically reviewing the relevant literature and quantifying how often drivers and the mechanisms linking drivers to ecosystem services are accounted for when assessing ecosystem service synergies and trade-offs. I also assess the types of drivers considered and the methods being used to identify trade-offs and synergies. I then discuss the implications for research into ecosystem service trade-offs and synergies so that policy and planning can be better informed.

2.3 METHODS

I conducted a systematic literature review of peer-reviewed articles using the ISI Web of Knowledge database and the search string: "ecosystem service*" AND ((synerg*) OR (trade-off* OR trade off* OR tradeoff*)). I limited the search to between 2005 and 2015 as previous ecosystem service reviews (Lee and Lautenbach 2016; Howe et al. 2014) found minimal literature before the publication of the Millennium Ecosystem Assessment (MA 2005). My search initially identified 1,298 scientific articles meeting the search criteria. I then screened the abstracts of each article and removed any articles that were not written in English, that were a review or conceptual paper, or that did not explicitly use qualitative or quantitative methods to identify ecosystem service trade-offs and synergies. Any articles where I was unable to determine if qualitative or quantitative methods were used from the abstract, were not removed, but were included in the next stage of screening. This process resulted in 240 articles. A second round of screening was then conducted by reading the full text of each article for relevancy, using the same criteria as in the first screening, which reduced the final number of articles to 113.

Papers were analysed using predefined questions and criteria (Table 2.1) drawing on previous ecosystem service reviews (Haase et al. 2014; Howe et al. 2014; Mouchet et al. 2014; Runting et al. 2017). Geographical data (study area) was extracted to identify any geographical patterns among the assessments of ecosystem service trade-offs and synergies, and the mechanisms and drivers identified. Data on the ecosystem services assessed in each study, and the relationships identified between services (synergy, trade-off, or no relationship) were then recorded. Because the names of specific ecosystem services were not consistent among the articles, I categorised each ecosystem service studied into “groups” using the Common International Classification of Ecosystem Services (CICES) V4.3 (see Table 2.1). This allowed for consistency in the identification of ecosystem service types among articles.

Table 2.1 Details of variables extracted from each article during the literature review.

Variables extracted	Categories
Study area	• Country(ies) where the study was located.
Consideration of the drivers of ecosystem service relationships	• Explicit: defined as identifying an ecosystem service relationship with potential drivers and causal pathways explicitly integrated into the assessment • Implicit: defined as identifying an ecosystem service relationship, with potential drivers identified or discussed, but not explicitly incorporating it into the assessment • No mention: defined as identifying an ecosystem service relationship, but not mentioning the driver or mechanisms leading to the synergy or trade-off
Driver identified	• Categories adapted from drivers of change for ecosystem services identified in (MA, 2005): Demographic; Socio-economic; Socio-political; Scientific and technological advances; Cultural and religious; Policy instruments; Land use/ Land cover change; Species introductions/ removals; Natural resource management; Harvest and resource demand; Climate change; Natural, physical and biological drivers (Nelson et al., 2006).
Method used to calculate ecosystem service trade-offs and synergies	• Correlation: measures the association between the supply of ecosystem services using correlation coefficients. • Overlap analysis: quantifies percentage of locations where two ecosystem services are provided at the same time. Trade-offs occur where one service is in high supply, and another is in low supply at different locations. Synergies occur where both service are simultaneously in high or low supply at different locations. • Ordination: multivariate analyses that order ecosystem service supply by values on multiple variables so that similar objects are near each other and dissimilar objects are farther from each other in ordination space. • ANOVA: Tests whether there are statistical differences between the means of different ecosystem services. • Regression: quantifies how the supply of an ecosystem service changes when the supply of one or more other ecosystem services change. Regression methods include general linear models, logistic models, structural equation models and path analysis. • Scenario analysis: a systematic method for developing alternative futures about the supply of ecosystem services.
Ecosystem services that were assessed	• Categorised based on CICES V4.3 Ecosystem Service Classifications, group level (http://cices.eu/): Biomass – nutrition (such as food production); Water (for human consumption); Biomass – materials (such as timber and plant based medicines); Water – materials (such as water used for industrial manufacturing); Biomass-based energy sources (such as biofuel); Mechanical energy (such as hydropower); mediation by biota (such as carbon storage and sequestration, absorption of pollutants); mediation by ecosystems (such as mediation of noise or smells, and filtration by ecosystems) ; mass flows (such as erosion control); liquid flows (such as flood mitigation); Gaseous/airflows (such as air ventilation); Lifecycle maintenance, habitat and gene pool protection (such as pollination; Pest and disease control (such a pest regulation); Soil formation and composition (such as soil fertility and nutrient storage); water conditions (such as regulation of water quality); Atmospheric composition and climate regulation (such as regulation of greenhouse gases); Physical and experiential interactions (such as hiking); Intellectual and representative interactions (such as education); spiritual and/or emblematic (such as spiritual identity); Other cultural outputs (such as enjoyment provided by existence of wild species) (Haines-Young and Potschin, 2013)
Ecosystem service relationship identified	• Trade-offs: one service increases, while the other decreases. • Synergy: two ecosystem services increase or decrease simultaneously.

The articles were categorised into three groups based on the extent to which they considered the drivers and mechanisms leading to trade-offs and synergies. These groups were: no mention, implicit, and explicit. *No mention* was defined as identifying an ecosystem service relationship, but not mentioning the driver or mechanisms leading to the synergy or trade-off. For example, Baral et al. (2013) identified a trade-off between forage production and water regulation, but did not mention what processes were driving the trade-off. *Implicit* was defined as identifying an ecosystem service relationship, and identifying or discussing potential drivers associated with the trade-off or synergy, and the pathways by which these drivers influence the relationship, but not explicitly quantifying the drivers or integrating them into the assessment. For example, Su et al. (2012) identified a trade-off between carbon storage and food production, and identified that human activity was associated with the presence of this trade-off, and therefore concluded it was a driver. However, that study did not explicitly identify the mechanistic links that explain how this driver influences this trade-off. *Explicit* consideration of drivers was defined as explicitly identifying the mechanistic pathways through which drivers influence ecosystem service relationships, and integrating this into the assessment. For example, Classen et al. (2014) used a controlled experimental design, by excluding and controlling variables in field plots, to explicitly identify pathways by which the presence of vertebrates drive a synergy between pollination, and therefore food production, and pest control. From the articles that either implicitly or explicitly identified the driver, I then extracted information on the type of driver being considered. Drivers were categorised into 12 groups that were adapted from the drivers of ecosystem service change outlined in the Millennium Ecosystem Assessment (MA 2005) (Table 2.1). The methodological approaches used by each article to identify trade-offs and synergies were also recorded to determine if there was a link between the methods applied and how drivers were considered. Methods were categorised into six classes: correlation, ordination, overlap analysis, ANOVA, regression, and scenario analysis.

I calculated the different spatial and temporal extents to which drivers were considered in the analyses (in total and per year), the frequency of the different methods used (in total and per year), the ecosystem services assessed, the trade-offs identified, the synergies identified, and the number of studies conducted in each country. A Pearson's chi-squared test was applied to assess whether the consideration of drivers and mechanisms was associated with the assessment method used.

2.4 RESULTS

2.4.1 Temporal and geographic patterns of ecosystem services assessed

Of the 113 articles examined, a total of 569 pairs of ecosystem service relationships were assessed (see Appendix C for a full list of the articles reviewed). Of these pairs, 254 were trade-offs, and 379 were synergies. The most common trade-off was between mediation by biota (ecosystem services provided by individual plants and animals) and biomass (food production) (n = 22). The most common synergy was between one type of biomass (e.g., crops) and another type of biomass (e.g., meat production) (n = 22), such as would occur between two different types of food production systems.

There was a rapid increase in the number of ecosystem service relationship articles published from 2005 to 2016. My literature review failed to find any articles published from 2005 and 2006 that focused on ecosystem service trade-offs and synergies. However, from 2007 onwards, the number of articles recorded increased with each consecutive year. Within these articles, there was a geographically wide distribution of case studies, with assessments of ecosystem service trade-offs and synergies in every continent, other than Antarctica. The three countries which equally had the highest number of assessments were China, Spain and the United Kingdom, which contained a combined total of 40% of all assessments identified (n = 15 per country). This was followed by the United States of America, which contained 12% of all assessments (n = 14).

2.4.2 Drivers and mechanisms assessed

Only 14.16% (n = 16) of the articles that I examined explicitly incorporated drivers of ecosystem service trade-offs or synergies into their assessment. However, a large proportion of articles (73.45%, n = 83) implicitly considered drivers. In general, articles that implicitly considered drivers either suggested possible drivers by identifying variables correlated with trade-offs and synergies, using statistical analyses (e.g., Ai et al. (2015)), or hypothesised potential drivers based on a review of the literature or field observations without the use of statistical analyses (e.g., Cohen-Shacham et al. (2011)), but did not explicitly incorporate mechanisms into their assessments. A small proportion of articles (12.39%, n = 14) made no mention of any drivers of the trade-offs or synergies. There was also a distinct trend over time in the way drivers and mechanisms have been considered in the assessment of ecosystem service synergies and trade-offs (Figure 2.2). Prior to 2008, while only two relevant articles were published, neither mentioned drivers in their assessments of ecosystem service synergies and trade-offs. However, from 2008 onwards there was a rapid increase in the proportion of articles that implicitly considered drivers. Also from 2008, a small number of articles began to explicitly incorporate drivers into their assessments. However, from 2010 onwards, while the number of papers implicitly considering drivers continued to grow, the number explicitly considering drivers did not (Figure 2.2).

Of the articles that did at least implicitly consider drivers of ecosystem service trade-offs and synergies (n = 97), there was a wide variety in the type of drivers identified (Figure 2.3). Physical and biological drivers were most commonly identified (31%, n = 30). This included drivers such as natural processes, biodiversity, hydrological regimes and net primary productivity. Other commonly considered drivers of ecosystem service relationships were related to policy instruments (28%, n = 27) and land management actions related to land use/land cover change (25%, n = 24). The effect of the introduction or removal of species (including invasive species) and cultural and religious practices as drivers of ecosystem service trade-offs and synergies were not considered by any of the articles examined.

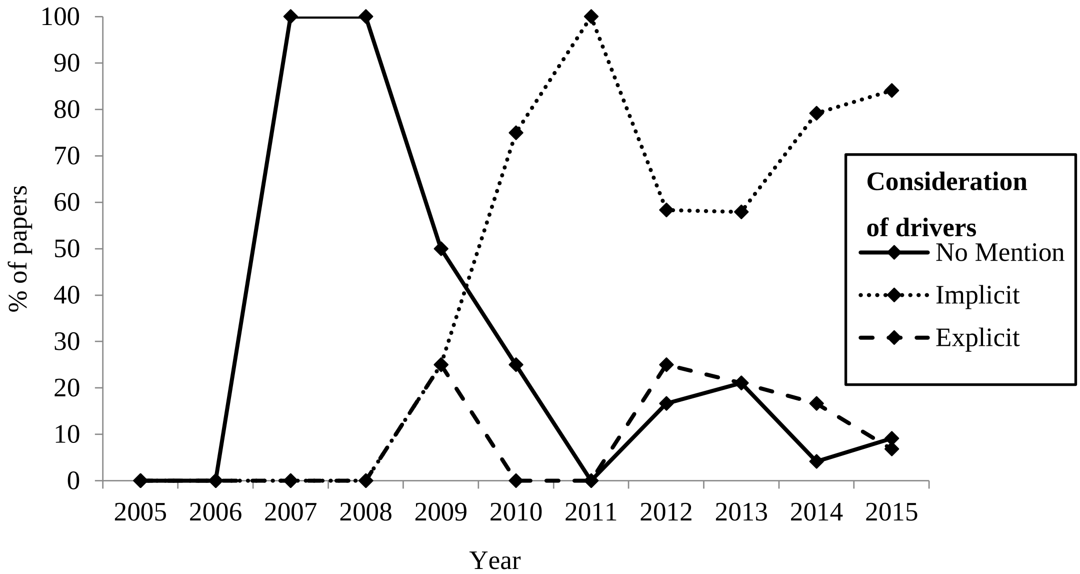

Figure 2.2. The percentage of papers considering the different categories of the drivers of ecosystem service trade-offs and synergies over time.

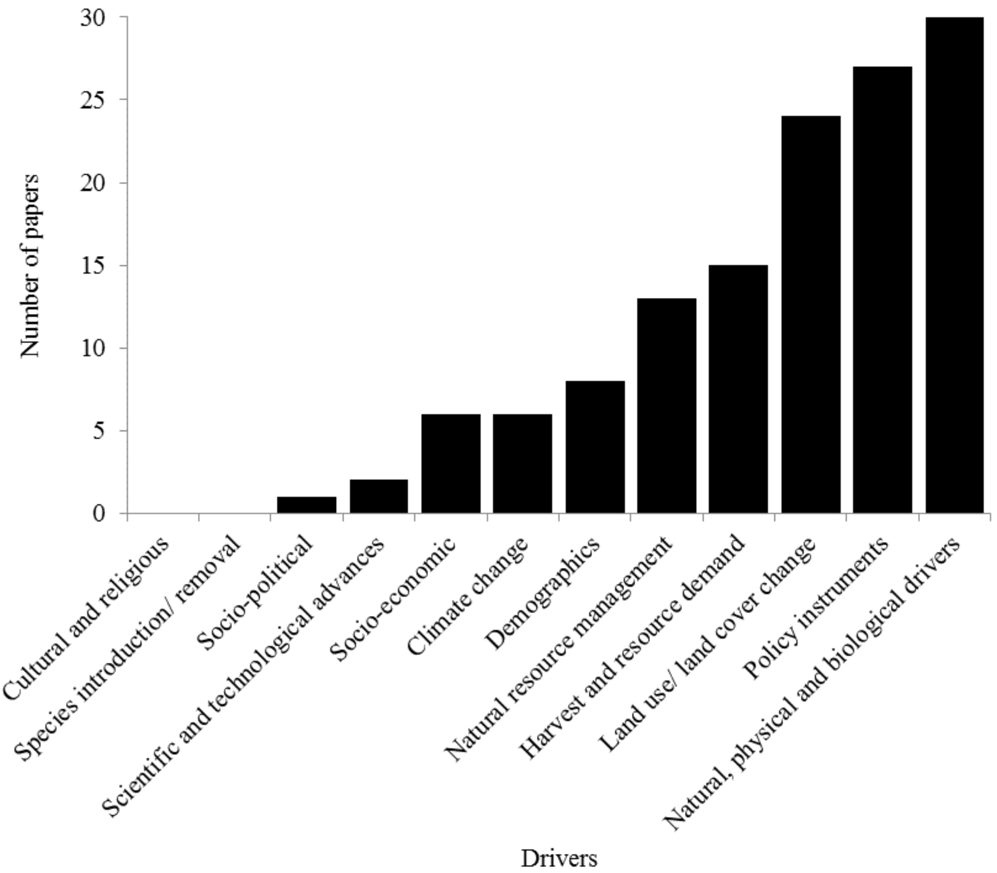

Figure 2.3. The frequency in which different drivers of ecosystem service trade-offs and synergies were identified in the examined articles.

2.4.3 Methods used to assess ecosystem service trade-offs and synergies

A variety of methods were used to assess trade-offs and synergies among ecosystem services (Figure 2.4). The most commonly utilised method was correlation (31%, n = 35). This method identifies ecosystem service trade-offs and synergies by simply quantifying the statistical association between pairs of ecosystem services. The second most utilised method was scenario analysis (27%, n = 31). This approach estimates ecosystem service trade-offs and synergies by projecting the provision of ecosystem services under different scenarios, and then using these values to identify services that change in the same direction (suggesting a synergy) or in opposite directions (suggesting a trade-off). Overlap analysis was also a common method (19%, n = 21). This employs a similar approach to correlation where trade-offs and synergies are identified based on spatial association or overlap. Regression methods were used less commonly (14%, n = 16), but consisted of a wide variety of different techniques, including generalised linear models, logistic regression models and structural equation models. The least common methods utilised were ordination (6%, n = 7) and ANOVA (3%, n = 3).

There was a significant association between the methods used to identify ecosystem service relationships and whether the drivers and mechanisms were considered explicitly, implicitly, or not at all (Chi-square test: $p \leq 0.001$, $\chi^2 = 38.59$, df = 12). Articles that explicitly incorporated drivers into the assessments tended to use scenario analysis, regression or ANOVA to identify trade-offs and synergies (Figure 2.4). On the other hand, articles where there was no mention of drivers tended to use correlation, ordination, overlap analysis, or regression methods.

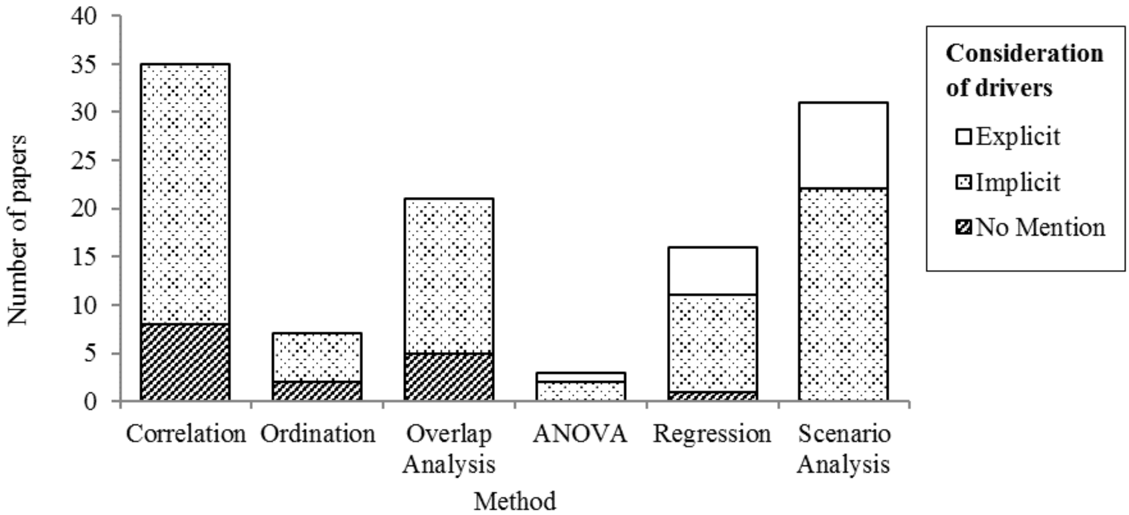

Figure 2.4. Frequency of the different types of methods used to identify ecosystem service trade-offs and synergies, and the number of articles within each method that implicitly, explicitly or did not mention the potential drivers of the relationships. ANOVA = Analysis of Variance.

2.5 DISCUSSION

Since the Millennium Ecosystem Assessment was published in 2005 (MA 2005), there has been a rapid increase in the understanding of the relationships between ecosystem services driven by the need to manage for multiple services (Bennett et al. 2009; Lee and Lautenbach 2016; Rodríguez et al. 2006). Despite recognition of the importance of drivers and mechanisms in determining synergies and trade-offs among ecosystem services (Bennett et al. 2009), my results show that few assessments are explicit about these processes. There is particularly a lack of focus on human drivers of ecosystem service relationships, such as cultural values, species management, and socio-political drivers. Nonetheless, there is evidence of improving recognition of the overall role of drivers and underlying mechanisms for assessing ecosystem service relationships. Future challenges lie in developing methods and data capable of a more explicit consideration of the drivers and mechanisms within socio-ecological systems relevant to ecosystem service provision. This will provide much greater confidence in predicting the consequences of policy interventions and other impacts on multifunctional outcomes.

Our review found that non-mechanistic methods are used more often than mechanistic methods to assess ecosystem service relationships. These non-mechanistic approaches, such as correlation and overlap analysis, have provided an important foundation for our understanding of ecosystem service relationships (Maes et al. 2012; Mouchet et al. 2014). However, they are unable to identify the causal drivers and mechanistic pathways that explain the relationships among services, limiting their ability to explicitly identify drivers (Iriondo et al. 2003; Sugihara et al. 2012). For example, one of the reviewed papers, Baral et al. (2013), determined that increasing land use intensification was spatially correlated with a decrease in carbon storage and regulation of water quality in the Lower Glenelg Basin, Australia, which suggests land use intensification is driving a negative synergy between carbon storage and water regulation. However, there may be other confounding variables, such technological advances and plant species diversity, which are affecting the provisioning of these services instead of land use intensification (George et al. 2012). Without explicit consideration of the mechanistic links between the potential drivers and ecosystem service relationships it is unclear what is actually driving the relationships between these services. Instead, typical mechanistic approaches that were used in the papers I reviewed, including experiments, scenario analyses and process-based models, were more capable of identifying and characterising the effect of causal drivers. For example, one paper (Lauf et al. 2014) used a simulation model — a process-based model capable of modelling the underlying mechanisms influencing ecosystem service provision — to identify the mechanistic pathway in which urbanisation drives a trade-off between energy production and food production in metropolitan Berlin, Germany. Furthermore, scenario analyses were often conducted using process based models that allow for the simulation of scenarios of the consequences of alternative drivers on synergies and trade-offs (Bagstad et al. 2013). These mechanistic approaches are able to quantify the strength of the mechanistic links between the ecosystem services and drivers to provide explicit information about trade-offs and synergies under different scenarios. This means they are much more likely to be able to effectively inform policy choices and avoid perverse outcomes (Lindenmayer et al. 2012).

Controlled experiments are the gold standard for identifying causal links between ecosystem variables (Schindler 1998). This approach was used in a small number of the reviewed articles, such as Classen et al. (2014), to identify how biodiversity drives a synergy between pest control and coffee production. However, the use of controlled experiments may be limited by the difficulty of controlling for multiple variables in the complex systems and at the broad spatial scales that are relevant for ecosystem services (Martinez-Harms et al. 2015; Sutherland 2006). An alternative when experiments are not possible is causal inference, that involves developing hypotheses about the causal links between variables while controlling for confounding factors in the sampling and statistical design (Law et al. 2017; Pearl 2009; Rubin 2005). However, no causal inference approaches were used by any of the reviewed articles. Process-based ecosystem service models were used in a number of the reviewed articles, including the ARIES and InVEST models (Balbi et al. 2015; Nelson et al. 2009). These types of models allow for the evaluation of the consequences of alternative scenarios (e.g., policy scenarios) for the relationships between ecosystem services, something that is not possible with purely correlative approaches. Therefore, there is great potential for applying these methods to identify general patterns in trade-offs and synergies under different drivers.

There was considerable variation in the number of papers focussing on each driver, but policy instruments were one of the most commonly studied type of driver. This suggests that there is a strong recognition of the importance of policy decisions for influencing trade-offs and synergies among ecosystem services. Properly dealing with the mechanisms underlying relationships that emerge from policy interventions would appear to be a particularly high priority (Ferraro and Hanauer 2014; Lindenmayer et al. 2012; Miteva et al. 2012). Human drivers, such as cultural and religious, socio-political, and scientific and technological drivers were the least considered drivers in my review. This may be due to a separation of the ecological and social sciences, which has led to a primary focus on the ecological processes underlying ecosystem service provision (Chan et al. 2012; Daniel et al. 2012; Liu et al. 2007). In recent years, there has been increased interest in assessing ecosystem services using a socio-ecological approach (Meacham et al., 2016). Applying more mechanistic approaches to identify ecosystem service relationships can aid in this due to their capacity to quantify the strength of linkages between social and environmental processes for ecosystem service provision (Spake et al. 2017), and identify trade-offs and synergies that are often ignored or misunderstood due to their social complexity (Daw et al. 2015).

The limited use of mechanistic approaches may simply reflect the often slow uptake of new methods, as they are often perceived as being risky, too difficult to implement, or because awareness of them is limited (Marra et al. 2003). Data availability may also play an important role in determining whether drivers and mechanisms can be incorporated into assessments of ecosystem service relationships. In many cases, the necessary data may not be available, and this will likely depend on the drivers or ecosystem services being assessed, the type of data required, the spatial scale, and available research budget (Bagstad et al. 2018; Spake et al. 2017). For example, drivers of cultural ecosystem service trade-offs and synergies may often require surveys and stakeholder interviews (Crouzat et al. 2016), which can be difficult to collect and incorporate into quantitative simulation models (Daniel et al. 2012). In this case, the integration of qualitative and quantitative data, through the use of mechanistic models can be a way forward to better reveal the relationships between ecosystem services (Martín-López et al. 2014). Therefore, when assessing ecosystem service trade-offs and synergies, I recommend that appropriate data collection to accommodate a mechanistic approach is identified early in the design phase to evaluate the data requirements and appropriate methodologies. This includes ensuring data is collected at an appropriate scale for analysing the mechanisms hypothesised as underpinning the ecosystem service relationships, and considering both social and ecological data requirements necessary to understand ecosystem service provisioning.

2.6 CONCLUSION

An incomplete understanding of ecosystem service trade-offs and synergies increases the likelihood of policy and management being ineffective, or being environmentally or financially costly (Degnbol and McCay 2007; Kremen 2005; Spake et al. 2017). A challenge for the assessment of ecosystem service trade-offs and synergies lies in developing research with a greater emphasis on drivers and the mechanisms that link drivers to ecosystem services. This requires consideration of drivers early in the design phase of research projects and encouraging greater uptake of methods, and collection of data, capable of identifying the mechanisms. A shift towards a more mechanistic understanding of the relationships between ecosystem services will result in better informed decisions for achieving sustainable and multifunctional landscapes.

CHAPTER 3

THE EFFECTS OF URBAN GREENSPACE CHARACTERISTICS AND SOCIO-DEMOGRAPHICS ON MULTIPLE CULTURAL ECOSYSTEM SERVICES

3.1 ABSTRACT

Urban parks provide many cultural ecosystem services that are essential for the wellbeing of residents, however we have little understanding of the key variables that determine the provisioning of these services. To effectively manage urban cultural ecosystem services it is necessary to determine which variables within urban landscapes are associated with the provisioning of these services. Here I disentangle the variables associated with four urban cultural ecosystem services (opportunities for exercise, nature interactions, relaxation and social interactions), within the urban park network of Brisbane, Australia, and provide insights about park management to simultaneously provide multiple cultural services. A spatially explicit survey of Brisbane residents provided empirical data on park visitations and use. This data was then used to develop location choice models to identify the key variables associated with park use. My results indicate that the variables affecting the likelihood of a park being visited for relaxation and nature interactions were quite similar, including the facilities present, vegetation structure of the park and the distant of the park from residents' homes, whereas exercise and social interactions was the only services influenced by the sociodemographic variables of urban residents. Furthermore, the degree to which variables were increasing or decreasing the rates at which parks were visited for multiple cultural services varied among the services, indicating that some variables were having a greater influence on some ecosystem services than others. By introducing management actions that target specific variables within urban parks it may be possible to facilitate the provision of multiple cultural ecosystem services. This could include increasing the facilities present and managing the tree cover within urban parks.

3.2 INTRODUCTION

Urban parks provide multiple cultural ecosystem services that are critical to the mental and physical wellbeing of urban residents (Bolund and Hunhammar 1999; Gómez-Baggethun and Barton 2013; Haase et al. 2014). This includes providing spaces for activities related to relaxation, physical exercise, social interactions, aesthetic appreciation, and other non-material benefits that people obtain from ecosystems (Brown et al. 2014; Daniel et al. 2012; Haase et al. 2014; MA 2005). With people increasingly living in urban areas (United Nations 2015), the demand for these cultural ecosystem services is growing rapidly (Andersson et al. 2014; Eigenbrod et al. 2011). To keep pace with this growing demand, urban parks need to be designed and managed so as to maximise the benefits and diversity of cultural ecosystem services they provide. This requires an understanding of the park design and socio-demographic factors driving park use for activities related to different cultural services in urban parks (Andersson et al. 2015). Though a number of studies have quantified the effect of park characteristics on park use and park visits (Bjerke et al. 2006; Giles-Corti et al. 2005; Jim and Chen 2006; McCormack et al. 2010; Shanahan et al. 2015), there remains limited knowledge on what drives the provision of multiple cultural ecosystem services simultaneously within urban parks (Dickinson and Hobbs 2017). This is an essential research gap to fill if urban planners and managers are to simultaneously improve the provisioning of multiple cultural services in cities.

There are a number of different characteristics of urban parks that can affect the activities and related cultural ecosystem services they provide, including spatial, vegetation and facility characteristics. Spatial variables, such as the location of parks can influence whether a park is used or not, as people are more likely to use parks closer to their homes (Cohen et al. 2007; Giles-Corti et al. 2005; McCormack et al. 2010). Vegetation variables, such as high tree cover and complexity in vegetation structure have also been found to be associated with longer visits for more nature-based activities, but shorter visits for physical exercise, therefore providing nature interactions but reducing opportunities for exercise (Bjerke et al. 2006; Shanahan et al. 2015). The size and type of parks can also be predictors of park use. Larger parks and linear parks are associated with more diversity in their use, increasing the number of activities (Brown et al. 2014; Cohen et al. 2010). Facilities present within urban parks can also have a large influence on how a park is used. The presence of activity equipment and paths are associated with increased park use for exercise (Humpel et al. 2002). People are also more inclined to use parks for social interactions that have a high number of public toilets, seating, barbeques, and playground equipment for children (McCormack et al. 2010).

The activities people use parks for is also heavily influenced by their socio-demographic characteristics (Kemperman and Timmermans 2008). For example, residents with a high income and a high level of education tend to be more frequent park users, and therefore are more likely to receive a wide array of cultural ecosystem services from parks (Cox et al. 2017; Shanahan et al. 2017). This could be due to people with these socio-demographic characteristics often living in higher socio-economic suburbs where parks are better maintained, aesthetically pleasing and safer (Leslie et al. 2010). The age of the park user is also associated with the activities they use parks for, with younger people more likely to use parks for activities related to exercise and older people more inclined to use parks for relaxation and nature interactions, possibly due to changes in wellbeing, leisure time, or a changing relationship to nature (Bjerke et al. 2006; Cox et al. 2017).

These relationships between visits and park and socio-demographic characteristics suggest that some of these variables could affect park visits for multiple activities, which lead to the provisioning of multiple urban cultural ecosystem services, in the same, or different, directions. Therefore, these variables could drive trade-offs and synergies to occur among the multiple cultural services provided within urban parks (Bennett et al. 2009). To effectively manage multiple cultural services within urban parks, it is necessary to first determine which park and socio-demographic variables promote synergies, and avoid trade-offs, to occur among the services. This requires simultaneously untangling the relationships between the use of parks for different activities that relate to multiple cultural ecosystem services, park characteristics and socio-demographic factors (Schipperijn et al. 2010).

In this chapter, I identify the variables that are related to the simultaneous provision of four cultural ecosystem services in urban parks: benefits from exercise, social interactions, relaxation, and nature interactions. Specifically, I aim to (i) identify the characteristics of urban parks and socio-demographic variables of urban residents that drive the use of parks for different cultural ecosystem services, and (ii) identify how the size and direction of the effects of these drivers varies among the urban cultural ecosystem services. Using Public Participation GIS (PPGIS) data on park use for different activities from Brisbane, Australia, I developed a spatially-explicit statistical model to identify the key park characteristics and visitor socio-demographic factors associated with each activity.

3.3 METHODS

I first reviewed the scientific literature to develop hypotheses about which visitor socio-demographic and park characteristics are most likely to affect park visitation for activities related to the four focal cultural services: exercise, nature interactions, relaxation and social interactions. I then collected data on the parks people are visiting to receive the different cultural services, and the socio-demographic variables of the visitors, using a participatory GIS survey. Data on the park variables were then collected through spatial and remote sensing data analysis. Statistical models were then developed to combine this data and identify which of these variables were most important in explaining the use of parks for the activities associated with each type of cultural service.

3.3.1 Study location

The Brisbane Local Governmental Area (LGA) in Queensland, Australia (Figure 3.1), occupies 1,380 km^2 and supported an estimated population of 1.1 million people in 2016 (ABS 2016b). I limited my study to all public greenspaces, from here on referred to as parks, across the Brisbane LGA, with park spatial data obtained from Brisbane City Council and Queensland State Government datasets (a summary of datasets used in this study is provided in Appendix D). A total of 2,872 parks covering 20,935 ha are present within Brisbane (Figure 3.1). The study area included one national park (D'Aguilar National Park), whose main function is nature conservation and has more limited public access. Due to the unique nature of this park, I excluded it from the study.

3.3.2 Identifying predictors of different cultural ecosystem services

In order to identify possible predictor variables of urban park use for different cultural ecosystem services, I conducted a literature review. Literature was sourced through a qualitative literature review using the Web of Science database. Database search terms included combinations of "urban", "ecosystem service", "urban green space" "urban park use", "recreation", "greenspace characteristics" and "culture". This semi-structured format was used to allow review flexibility and the ability to fully explore the literature (Dickinson and Hobbs 2017). From each paper, I then extracted data on the urban park activities and predictor variables studied, and whether these variables positively or negatively affected activity levels. Activities were categorised into one of four cultural ecosystem services categories: opportunities for exercise, nature interactions, relaxation, and social interactions (Table 3.1). Using this data, I identified which park and socio-demographic factors are likely the most important predictors of the frequency of park visits for each of the four services and divided these variables into four categories based on their characteristics: spatial, socio-demographic, environmental, and park facility variables.

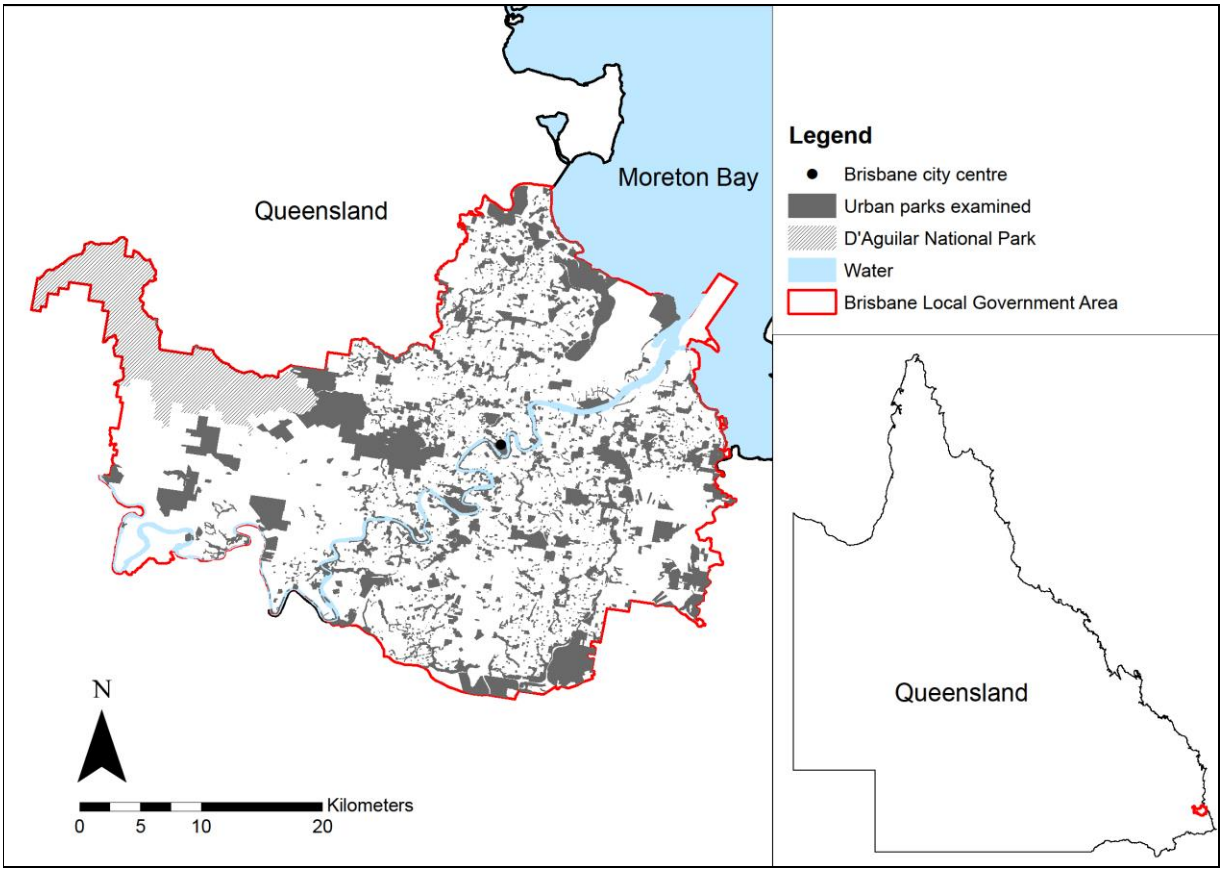

Figure 3.1 Map of the Brisbane Local Governmental Area (LGA) showing the urban parks considered in this study. D’Aguilar National Park was not considered in this study.

Table 3.1 Explanatory variables of cultural ecosystem service provision identified through a literature review and developed into hypotheses to explain which predictor variables (drivers) were likely to have the largest influence on park use for activities related to the four cultural ecosystem services. The shaded boxes denote that a significant relationship between the driver and ecosystem service was recorded in the literature, and the +/- symbols denote whether it was recorded as positive or negative relationship, or both.

Predictor category	Predictor variable	Predictor variable definition	Effect on urban park cultural ecosystem services provision				Reference examples
			Exercise	Nature Interactions	Relaxation	Social Interactions	
Spatial	Distance from home	Distance of the park from the visitor's home	-	-	-	-	• Cohen et al. (2007) • Rossi et al. (2015)
Environmental	Tree cover	The amount of tree cover present within the park	-	+			• Shanahan et al. (2017) • Shanahan et al. (2015) • Bjerke et al. (2006) • Cohen et al. (2010) • Giles-Corti et al. (2005) • Brown et al. (2014)
	Grass cover	The amount of grass cover within the park	+			+	
	Foliage height diversity	The diversity in the number of vegetation strata present within areas of tree cover	-	+			
	Size of park	Area occupied by the park	+	+	+	+	
	Shape of park	How uncompact (square) the perimeter of the park is	+				
Facilities	Amenities	Toilet blocks, benches, tables and shade structures	+		+	+	• Nordh and Østby (2013) • Peters et al. (2010) • McCormack et al. (2010) • Shores and West (2010) • Schipperijn et al. (2013) • Humpel et al. (2002)
	Accessibility	Length of pathways	+	+			
	Exercise equipment	Fitness equipment with cardio and resistance features	+				
	Animal facilities	Off-leash dog areas	+			+	
	Children's play equipment	Playgrounds	+		-	+	
Socio-demographic	Gender	Park visitor identifies as male				+/-	• Ching-hua (2005) • Cox et al. (2017) • Bjerke et al. (2006) • Chiesura (2004)
	Income	The amount a park visitor earns	+	+	+	+	
	Age	The age of the park visitor	-				
	Lifecycle	Stage of life of the park visitor. Stage of life is indicated by a combination of marriage status, presence of children and age of children	-			-	
	Education	The level of education the park visitor has obtained	+	+	+	+	• Shanahan et al. (2017)

3.3.3 PPGIS survey

I conducted a Participatory Geographic Information Systems (PPGIS) survey to determine the frequency of park visits by Brisbane residents for the different cultural ecosystem services, and to collect data on the socio-demographic characteristics of park visitors (see Appendix D to view the survey). PPGIS is a field within geographic information science that focuses on the ways the public uses various forms of geospatial technologies to participate in public processes, such as mapping and decision making (Brown and Fagerholm 2015; Brown and Kyttä 2014). This approach was chosen as it is capable of capturing the locations of park use in a spatially explicit way that can then be related to park characteristics.

The PPGIS survey was conducted from October 2016 to January 2017 with participants randomly selected from across Brisbane using residential mailing addresses for the Brisbane Local Governmental Area obtained from a commercial database (yell123.com). Survey invitations, including a web link to the PPGIS survey interface, were sent to a sample of 7,500 addresses selected from this database, stratified across suburbs with weighting proportional to the area of each suburb. Participants were awarded a $10 gift voucher for participating, as an incentive. At the website, participants were presented with a customised Google® map interface of the Brisbane LGA, highlighting all of the urban parks. They were then instructed to drag and drop different digital markers onto the parks representing activities that they had conducted at these locations within the past two weeks. Participants could choose from eleven different activity markers that were related to the four different cultural ecosystem services (Table 3.2). Once the mapping activity was complete, participants were asked a number of basic socio-demographic questions. These were used to identify bias in the survey toward particular socio-demographic groups, and to determine whether use of parks were related to socio-demographic characteristics of the survey participants (see Appendix D for the complete list of questions). Only markers that were placed inside parks by participants during the PPGIS survey were used in further analyses. To test how representative the survey population was, I compared the survey participant demographic variables to the Australian Bureau of Statistics census data for the Brisbane LGA (ABS 2016b).

Table 3.2 List of activities, number of markers placed by survey participants, and the cultural ecosystem services they represent in the PPGIS survey.

Activity	Cultural ecosystem service component	Reference
Walking	Exercise	Brown et al. (2014)
Running/jogging	Exercise	Irvine et al. (2013)
Cycling	Exercise	Irvine et al. (2013)
Play sport	Exercise	Irvine et al. (2013)
Dog walking	Exercise	Cutt et al. (2008)
Use exercise equipment	Exercise	Bedimo-Rung et al. (2005)
Boot camp	Exercise	Brown et al. (2014)
Observe nature	Nature interactions	Irvine et al. (2013)
Resting/sitting	Relaxation	Bedimo-Rung et al. (2005)
Social activities	Social interactions	Irvine et al. (2013)
Supervise children	Social interactions	Irvine et al. (2013)

3.3.4 Explanatory variables

The value of each explanatory variable for each park in Brisbane was then calculated to use in the development of the statistical models (Table 3.3). Due to some participants choosing to not provide information on age, income, life cycle and education, there was missing data for these factors for some of the survey respondents. For age, life cycle and education, I deleted all participants with missing data, as there were only a few of these cases (13 participants). For income, there was a large amount of missing data (139 participants). I therefore used data imputation to fill in the missing data for income using the approach described in Outhwaite and Turner (2007). A linear regression model was developed using Age, Gender and Education to estimate the income of a survey participant ($R^2 = 0.08$, $P < 0.05$). This model had a low R^2, but models containing social data often have low R^2 values due to the large number of covariates affecting the data (Abelson 1985). However, the model was significant and deemed to be a better alternative than removing participants with missing income data.

I determined the distance between participant's home addresses and parks by measuring the distance from each participant's postal address to each park via the Brisbane road network. This was completed using the Network Analysis tool in ArcGIS 10.3.1 with a spatial dataset of Brisbane's road network sourced from the Queensland Department of Natural Resources and Mines (2017).

Spatially explicit maps for tree cover, grass cover and foliage height diversity (FHD), a measure of the evenness of vegetation density across vertical strata, with higher values indicating greater evenness (MacArthur and MacArthur 1961), were derived from existing LiDAR and other high resolution remote sensing imagery (Caynes et al. 2016; Mitchell et al. 2016). Tree cover for each park was determined at a 5 x 5 m resolution from the mean foliage projective cover, the vertically projected percentage cover of vegetation of all strata (Caynes et al. 2016). This was sourced from Brisbane City Council and derived from a 2009 LiDAR data set using methods described in Armston et al. (2009). Tree cover was mapped by identifying areas where foliage projective cover was greater than zero and greater than 2 m in height, and I used this to calculate the proportion of tree cover in each park. Using the same LiDAR data set, I also calculated the proportion of grass cover in each park and the average FHD for each park following Caynes et al. (2016).

I calculated the shape of each park using the Shape Index as defined in McGarigal et al. (2002):

$$SHAPE = \frac{P_{ij}}{\min P_{ij}}$$

Where P_{ij} is the perimeter of the park *ij*, and min P_{ij} is the minimum perimeter of park *ij* possible if the park area was rearranged to make a maximally compact shape (McGarigal et al. 2002). A value of 1 indicates that the park is maximally compact and as the value increases from one it indicates that the shape of the park is increasingly irregular.

Data on the facilities present within each park was obtained from Brisbane City Council (Brisbane City Council 2015) and included amenities, such as toilets and benches; exercise facilities, such as exercise equipment; access facilities, such as paths and roads; facilities for animals, such as dog litterbags and off leash dog areas; and children's play facilities. Previous studies have shown that the presence of facilities within parks are more important than the number of facilities in attracting people to parks for various activities (Edwards et al. 2015; Kaczynski et al. 2014; Potwarka et al. 2008). Therefore, for this analysis, the number of each type of facility was standardised into presence/absence data for each park.

All continuous predictor variables were standardised using the z-transformation prior to analysis and a test for collinearity between predictor variables was also conducted based on Spearman's Rank Correlation Coefficient. As suggested in Dormann et al. (2013), correlations between pairs higher than ± 0.7 indicated high collinearity and were removed.

Table 3.3 List of the indicators used to measure each predictor variable hypothesised as influencing the use of urban parks for different activities, to receive one of the four cultural ecosystem services. The data sourced for each indicator is also listed.

Driver category	**Variables assessed**	**Indicator**	**Data source**
Socio-demographic	Gender	Male; Female	Social survey
	Income	Average weekly income	Social survey
	Age	Age (years)	Social survey
	Life cycle	Young single; mature single; young couple/no children; mature couple/no children; young family; middle family; senior family; older couple/no children living at home	Social survey
	Education	High School certificate, College Diploma (TAFE), University Degree	Social survey
Spatial	Distance from home	Distance from house to nearest edge of greenspace boundary (m)	Social survey, spatial data
Environmental	Tree cover	Proportion park with tree cover	LiDAR data
	Grass cover	Proportion of park with grass cover	Land cover map/ LiDAR data
	Foliage height Diversity	Average foliage height diversity measure for park	LiDAR data, with method outlined in Caynes et al. (2016)
	Size of park	Area of park (m^2)	Spatial data
	Shape of park	Perimeter of park / minimum perimeter for maximally compact shape	Spatial data/ FRAGSTATS (McGarigel et al. 2012)
Facilities	Amenities	Presence/absence of benches, shade devices, barbeques, toilets within park	Brisbane City Council (2015)
	Access facilities	Presence/absence of paths within park	Brisbane City Council (2015)
	Exercise facilities	Presence/absence of exercise equipment within park	Brisbane City Council (2015)
	Animal facilities	Presence/absence of off-leash dog zones within park	Brisbane City Council (2015)
	Play facilities	Presence/absence of child play equipment within park	Brisbane City Council (2015)

3.3.5 Data analysis

I used generalised linear models to model the number of park visits for each ecosystem service as functions of the park characteristics, socio-demographic variables, and distance to park. While discrete choice models may be the preferred model for this type of analysis (Hanley et al. 2001), the large number of respondents and alternative choices (i.e., alternative parks) precluded the use of discrete choice models for computational reasons. However, Schmidheiny and Brülhart (2011) show that, for location choice models, generalised linear models provide similar parameter estimates to discrete choice models. Similar types of analysis for species habitat selection also commonly use generalised linear models for modelling habitat choice (Segurado and Araújo 2004). As the response variable (number of park visits) contained a large number of zeros, I used zero-inflated Poisson models (Martin et al. 2005). I used a zero inflated model that consists of a mixture of two distributions: a binomial distribution representing the probability that a park is visited at least once, and a Poisson distribution representing the number of visits given the park is visited at least once (Wenger and Freeman 2008). To account for dependence between multiple reported visits from the same individual, I include a random intercept effect for individuals in the model.

I then constructed alternative regression models that included different combinations of the spatial, environmental, facilities, and socio-demographic variables on the binomial and count components of the model. All models included the distance variable in both the binomial and count components because there is already strong evidence in the literature that this is almost always important (see Table 3.1). I then considered alternative models consisting of all combinations of groups of environmental, facilities and socio-demographic variables included in the binomial component, resulting in eight alternative models (Table 3.4). I included variables only as linear effects and did not consider interactions. I chose to only include the key explanatory variables of interest in the binomial component because I hypothesised that the largest effects would be on the propensity to visit a park or not and because models failed to converge when variables were included in both the binomial and count components. Each of the models were fitted to the data and ranked based on their Akaike's Information Criteria (AIC) to identify the most parsimonious models. The goodness of fit of the most parsimonious model for each cultural service was assessed using quantile-quantile (Q-Q) plots. All data analyses were performed using the R statistical program (version 3.4.0), with the zero-inflated Poisson models developed using the glmmTMB package (Brooks et al. 2017), and the Q-Q plots developed using the DHARMa package (Hartig 2018).

Table 3.4 The eight alternative models tested. Each model consists of a different combination of predictors; environmental, spatial, facility, and socio-demographic predictors. For each model, the predictor variables were included within the binomial part of the model, except for the "distance from home" variable which was included in both the count and binomial parts of every model.

Model		Predictor variables included
1.	Environmental + Spatial	Distance from home + Grass cover + Tree cover + Foliage Height Diversity + Shape of park + Size of park
2.	Facilities + Spatial	Distance from home + Amenities + Play facilities + Access facilities + Active facilities + Animal Facilities
3.	Socio-demographic + Spatial	Distance from home + Gender + Education + Income + Age + Life cycle
4.	Environmental + Facilities + Spatial	Distance from home + Grass cover + Tree cover + Foliage Height Diversity + Shape of park + Size of park + Amenities + Play facilities + Access facilities + Active facilities + Animal Facilities
5.	Environmental + Socio-demographic + Spatial	Distance from home + Grass cover + Tree cover + Foliage Height Diversity + Shape of park + Size of park + Gender + Education + Income + Age + Life cycle
6.	Facilities + Socio-demographic + Spatial	Distance from home + Amenities + Play facilities + Access facilities + Active facilities + Animal Facilities+ Gender + Education + Income + Age + Life cycle
7.	Environmental + Facilities + Socio-demographic + Spatial	Distance from home + Grass cover + Tree cover + Foliage Height Diversity + Shape of park + Size of park + Amenities + Play facilities + Access facilities + Active facilities + Animal Facilities+ Gender + Education + Income + Age + Life cycle
8.	Spatial only	Distance from home

3.4 RESULTS

A total of 474 residents participated in the PPGIS survey which was a response rate of 7% (474/7,500). Other internet-based PPGIS surveys have reported response rates of around 10% (Pocewicz et al. 2012). The participants mapped the location of 2,239 activities across Brisbane's parks with 1,654 related to exercise, 168 related to nature interactions, 117 related to relaxation, and 300 related to social interactions (Table 3.5). The most commonly placed activity marker was "Walking" (n = 796), and the least placed marker was "Boot Camp" (n = 25). There was a sampling bias in the survey population towards people who were older, more likely to have a bachelors/postgraduate degree, and have an income of $2,000 per week or more (Table 3.6).

Table 3.5 A summary of the number of markers placed by the survey participants for each activity.

Activity	Cultural ecosystem service component	Number of markers placed in the PPGIS survey
Walking	Exercise	796
Running/jogging	Exercise	185
Cycling	Exercise	253
Play sport	Exercise	77
Dog walking	Exercise	282
Use exercise equipment	Exercise	36
Boot camp	Exercise	25
Observe nature	Nature interactions	168
Resting/sitting	Relaxation	117
Social activities	Social interactions	114
Supervise children	Social interactions	186

Table 3.6 Summary of survey participant statistics compared to the summary statistics of the 2016 Australian Census for the Brisbane Local Governmental Area.

	Census (ABS, 2016b)	Survey participants
Number of survey participants	-	474
Number of markers placed	-	2239
Gender		
Male (%)	49.2	56.0
Female (%)	50.8	44.0
Age in years (mean)	35.2	53.0
Highest education level attained		
High School (%)	39.3	43.8
TAFE/College Diploma (%)	30.6	5.0
University degree (%)	20.8	51.2
Income (weekly)		
$2,000 or more (%)	11.2	23.0
Life cycle		
Couple with children (%)	44.2	47.9

For each of the cultural ecosystem services there was one model that was clearly the most parsimonious (AIC weights > 0.85) indicating little model uncertainty (Table 3.7). The Q-Q plots for each of these parsimonious models identified a strong linear regression between the quantiles of the observed and predicted model data (see Appendix D). This suggests the standardised residuals have a uniform distribution and there is minimal misspecification in these models. For activities related to relaxation and nature interactions, the models including environmental and facility variables were the most parsimonious models of park visitation (Table 3.7). For activities related to exercise and social interactions the models including socio-demographic, environmental, and facility variables were the most parsimonious models of park visitation.

Table 3.7 AIC values and weights for each park visitation model developed for activities related to the four cultural ecosystem services. E = environmental variables; D = spatial variable (distance from home variable); F = facility variables; SD = socio-demographic variables. The variable "distance from home" was included in all models. Values in grey denote the most parsimonious models.

	Exercise		Nature interactions		Relaxation		Social interactions	
Model	**AIC**	**AIC weight**	**AIC**	**AIC weight**	**AIC**	**AIC weight**	**AIC**	**AIC weight**
1. E + D	14446.2	0	2040	0	1758.7	0	3478.2	0
2. F + D	14446.3	0	2158	0	1739.2	0	3297.7	0
3. SD + D	15658.9	0	2273	0	1855.7	0	3586.1	0
4. E + F + D	13766.7	0.148	1989.9	0.966	1686.1	0.921	3215.7	0
5. E + SD + D	14446.2	0	2175.9	0	1763.8	0	3436.1	0
6. F + SD + D	14442.3	0	2166.1	0	1744.7	0	3258.7	0
7. E + F + SD + D	13763.2	0.852	1996.6	0.034	1691	0.0794	3175.5	0.999
8. D	15659.7	0	2264.9	0	1850.2	2.139E-36	3627.1	0

Significant socio-demographic and park variable coefficients ($p \leq 0.05$) within the best models varied among all four cultural ecosystem services (Figure 3.2). For all four cultural ecosystem services, the “distance from home” variable was negative in both the count and binomial parts of the model. However, the magnitude of the distance from home effect differed among the services, with nature interactions and exercise showing the greatest negative effect in both the binomial and count sections of the models. This suggests that people may be prepared to travel less far to use parks for nature interactions and exercise than they are prepared to do for social interactions and relaxation.

Of the environmental predictor variables, the size and shape of the park both had significant positive impacts on all four services suggesting that larger and more linear parks are used more frequently than smaller more compact parks. The proportion of tree cover present was negatively related to the chance of using a park for exercise, relaxation and social interactions, but had no significant association with nature interactions. However, FHD did have a positive association with the chance of visiting a park for activities related to nature interactions and exercise, but with the FHD effect for activities related to nature interactions being larger than for exercise. This suggests that, as vertical evenness of vegetation density increases (e.g., due to increased mid- and under-story vegetation), the chance of a person visiting a park for either nature interactions or exercise increases, but particularly for nature interactions. The proportion of grass cover had only a significant association for social interactions and the relationship was negative.

Of the facilities present within parks, access, via footpaths and tracks, was associated with a higher chance of using a park for all four cultural ecosystem services. Children’s play equipment had a positive association with the chance of visiting a park for social interactions, exercise and nature interactions, but not relaxation. However, for social interactions, play equipment had the largest coefficient suggesting this is a key factor associated with social interactions. Presence of exercise equipment was associated with increased chance of someone visiting a park for activities related to social interactions and exercise. Presence of amenities (benches, shade devices, BBQs, toilets) was associated with an increased chance of someone visiting a park for activities related to exercise and relaxation. However, for activities related to relaxation, the coefficient for amenities was higher than that for exercise.

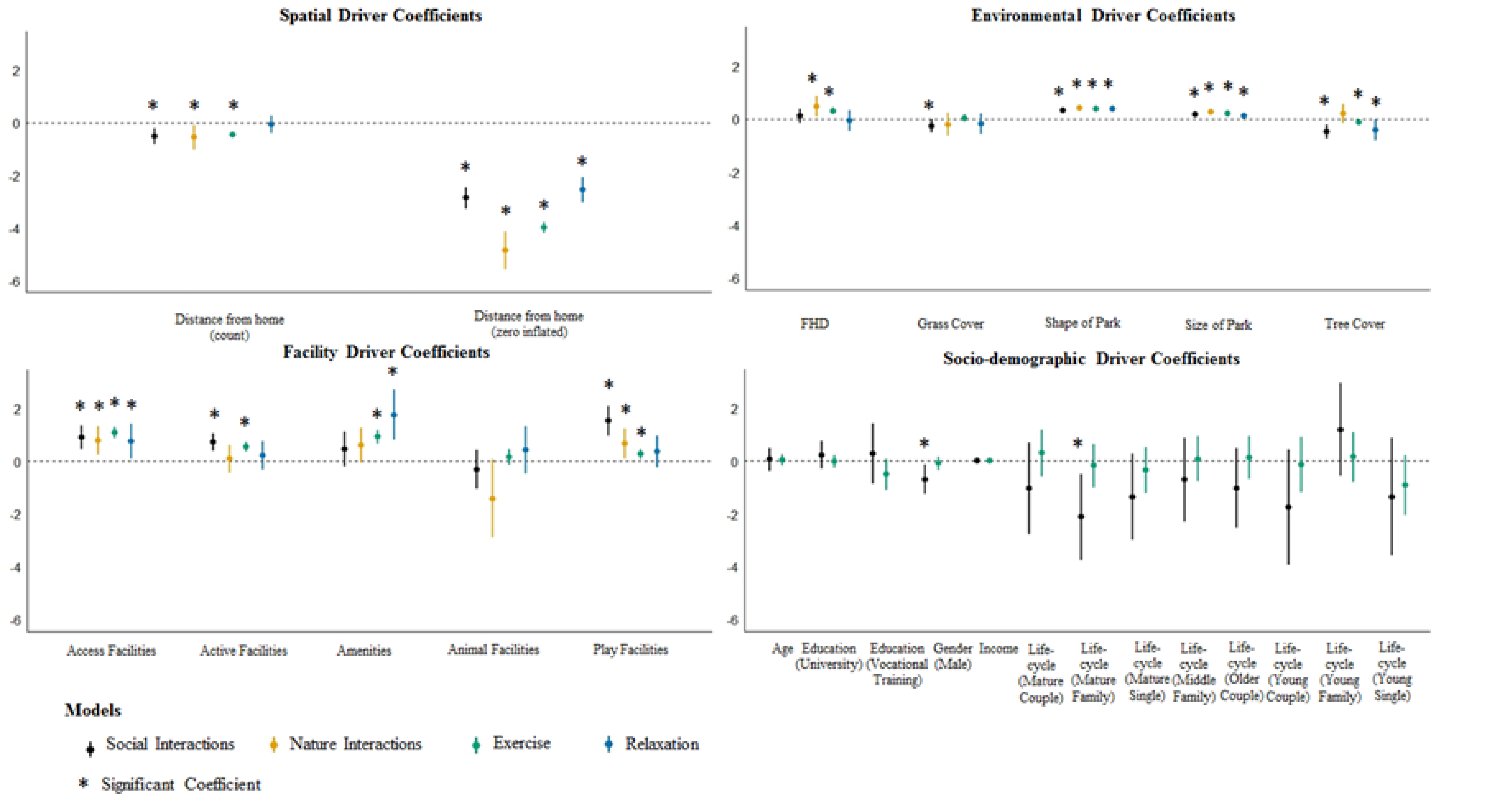

Figure 3.2 Coefficients of the variables within the count ("Distance from home") and binomial (all variables) components of the most parsimonious models for the four assessed cultural ecosystem services. The vertical lines denote the 95% confidence intervals for each coefficient. Variables with confidence intervals which did not overlap zero were considered significant. FHD = foliage height diversity

There were few individual socio-demographic variables significantly associated with the use of parks for any of the ecosystem services. For activities related to social interactions, only gender and the life stage of the person had a significant association with the chance of visiting a park. Males and people from mature families visited parks less for social interactions than females and people from other life stages. For exercise, no socio-demographic variables were individually significant. However, the confidence intervals for the socio-demographic variables were large, suggesting there are other variables affecting park visits for different activities that were not captured in this survey (Figure 3.2).

3.5 DISCUSSION

While many studies have focused on predictors of park use (see McCormack et al. 2010), this study provides a comparison of predictors for a variety of different activities within the same system, and interprets this in terms of cultural ecosystem service provision. Understanding this is critical if we are to effectively manage parks for multiple cultural services. I show that environmental, facility, spatial and socio-demographic characteristics of urban parks are all key factors determining which parks people visit for different cultural ecosystem services. There are many similarities with how these variables affect the propensity to use parks for different cultural ecosystem services. However, the degree to which these variables affect park use varies among different activities and this influences the ability of park management to simultaneously maximise multiple cultural services. Further, by developing an understanding of the associations between use of parks for different services and the characteristics of parks and socio-demographic characteristics of residents, we can better understand what variables are driving trade-offs and synergies among urban cultural services. This will enable us to identify the key variables to manage that will generate synergies among ecosystem services, effectively managing multiple cultural services simultaneously.

The most parsimonious models for all four ecosystem services included the spatial, environmental and facility variables. As these three groups of variables include only characteristics of the park itself, rather than of the people using the park, this highlights that the location and characteristics of the park is important in the provisioning of all four ecosystem services, and solely important for the provisioning of nature interactions and relaxation. This supports the findings of Bertram and Rehdanz (2015) who found that the cultural ecosystem service provisioning within urban parks across Europe is related to the quality and design of the park. The only parsimonious models to include the socio-demographic variables were the social interactions and exercise models. Therefore, the characteristics of the beneficiaries of these cultural services (the park visitors) are important to the provision of these services. Previous studies have found that park visits for activities related to these services are often conducted by particular social groups making socio-demographic characteristics important to ecosystem service provisioning (Shan 2014). However, these results could also be due to social inequality in the provisioning of exercise and social interactions, with people of certain socio-demographic characteristics having better access to parks suitable for exercise and social interactions (Rigolan 2016).

The predictor variables significantly associated with park use were similar across many of the activities related to the different cultural ecosystem services. Tree cover had a negative impact on the chance of a park being visited for activities related to social interactions, relaxation and exercise, but had no influence on activities related to nature interactions. Instead, this service was related to foliage height diversity (FHD). The negative relationship between tree cover and park activities could be attributed to a number of reasons. Jorgenson et al. (2002) found that parks users felt unsafe in urban parks with high tree cover due to reduced light and inability to see far distances, and tree cover also acts as an obstacle for sporting activities (Cohen et al. 2007). It is unsurprising that nature interactions is positively associated with foliage height diversity as this is often associated in higher diversity in plant and bird species (Erdelen 1984; McKinney 2008). The size and linearity of parks had positive relationships with park use for all activities. This relationship is most likely due to large parks being able to provide more functional zones for a wider variety of activities (Lam et al. 2005), and linear parks have a much larger perimeter than compact parks, improving access for a larger number of people (Sister et al. 2010). The presence of facilities within greenspaces acted as a positive driver on activities related to all the cultural ecosystem services. Providing facilities within greenspaces provide convenience to park visitors, attracting a more diverse range of park visitors who engage in a wider range of activities (McCormack et al. 2010).

The results of this study have implications for the effective management of multiple urban cultural services. Though the provisioning of the cultural ecosystem services are often driven by the same predictor variables, the size of the predictor coefficients changed in each of the ecosystem service models. This demonstrates that the degree of influence of each predictor variable on each ecosystem service is not the same. Understanding the influence of multiple variables on the rate of park visitation for exercise, relaxation, nature interactions and social interactions can help determine which variables should be targeted to increase the provisioning of multiple cultural ecosystem services, or a single service, with minimum cost (Gómez-Baggethun and Barton 2013). For example, my results show that the chance of visiting a park for activities related to exercise increases when the presence of both access and children's play facilities increase. However, due to the larger size of the coefficient observed for access facilities than that for children's play equipment in the model, the chance of a park being visited for activities related to exercise will increase at a faster rate if access facilities are increased than if children play facilities are increased within the park. Therefore, management actions targeting the variables with a high coefficient for one service is suitable when focusing on a single service, but the most effective variable to target when managing multiple services may differ and become a more complex decision.

While many studies have focused on identifying variables that have similar effects on multiple ecosystem services (Raudsepp-Hearne et al. 2010; Renard et al. 2015), the degree of influence of the variables on each individual ecosystem service is often not considered (Kremer et al. 2016). My models allow decision makers to identify the variables to focus management actions on to tailor urban parks to deliver multiple cultural ecosystem services that are demanded for by local residents. As well as improving cultural ecosystem service provisioning in existing parks, this knowledge can also be used to plan new urban park developments that can accommodate the demand for cultural ecosystem services based on projected urban resident population growth (Jannson 2013; Andersson et al. 2014). This forward planning of cultural ecosystem service delivery in both future and existing parks is crucial for maintaining the physical and mental wellbeing of urban residents.

To improve ecosystem service provisioning of urban parks, it is also necessary to understand the factors that prevent people from visiting parks entirely to receive cultural ecosystem services. To achieve this requires information on the people who choose not to visit parks, and to link this to socio-demographic and park characteristics. The ecosystem service models developed in this study estimate the frequency of park visits, given a person visits a park at least once. Therefore, this study only assesses the likelihood of a person visiting a specific park, not the likelihood of person using any park at all. To effectively introduce management actions that increase the provisioning of cultural services, it is necessary to understand the variables that prevent people from visiting any park at all to receive these services (Lin et al. 2014). These variables could include how much access a person has to private greenspace, the socio-demographic characteristics of the person and the spatial location of the person (Lin et al. 2014; Rossi et al. 2015). Further research is required assess these variables to better understand the variables affecting peoples' use of parks for cultural ecosystem services. Future research should also focus on identifying the variables generating trade-offs and synergies occurring not just between the cultural ecosystem services, but also with other regulatory and provisioning ecosystem services that are important for human wellbeing in urban areas, such as noise reduction and air quality regulation (Haase et al. 2014). This will ensure management actions target variables within parks that do not decrease important regulating and provisioning services provided by parks (Jim and Chen 2008; Mitchell et al. 2018; Nowak and Crane 2002; Nowak et al. 2006).

3.6 CONCLUSION

The importance of urban greenspace characteristics on influencing visits for different activities has been well documented (McCormack et al. 2010). However, few studies have linked these characteristics to multiple activities that provide a variety of cultural ecosystem services (Luederitz et al. 2015). This knowledge gap makes it difficult to implement management and policies that will ensure urban parks provide multiple cultural ecosystem services sustainably. By using a spatially explicit approach, my study linked urban park characteristics to the delivery of multiple cultural ecosystem services. My study demonstrates that management actions focused on managing tree cover and the number of facilities present within urban parks could increase provisioning of multiple cultural ecosystem services within these parks. This information can support decision makers in designing urban parks to accommodate the cultural ecosystem service demand for a growing urban population (United Nations 2015).

CHAPTER 4

URBAN ECOSYSTEM SERVICE TRADE-OFFS AND SYNERGIES UNDER DIFFERENT MANAGEMENT SCENARIOS

4.1 ABSTRACT

Urban greenspaces are becoming increasingly important sites for ecosystem service delivery. Managing these spaces to maintain the provision of multiple ecosystem services is challenging due to the positive (synergistic) and negative (trade-off) relationships that exist between services. To prevent trade-offs from occurring, it is necessary to first identify how different management actions affect the relationships between ecosystem services. However, there is currently limited knowledge on how management actions affect ecosystem service relationships. Here, I assess the relationships occurring among five ecosystem services (carbon storage, and opportunities for exercise, nature interactions, relaxation and social interactions) under three potential urban revegetation management scenarios, which focus on altering mid storey vegetation and tree cover, across the public greenspace network of Brisbane, Australia. To determine whether the ecosystem service trade-offs and synergies occurring change under the different management actions, I identified these relationship using two different methods; a spatial correlation that can calculate ecosystem service relationships presently occurring spatially across the landscape, and a scenario analysis that can calculate ecosystem service relationships occurring over time as revegetation actions are introduced. The spatial correlation method only identified synergies occurring spatially between the cultural ecosystem services. However, the scenario analysis identified both trade-offs and synergies between these ecosystem services under the management scenarios that involve increasing tree cover. My results suggest that ecosystem service relationships can change as different management actions are implemented. I recommend greater uptake of methods capable of identifying the changes in ecosystem service relationships under different management actions when assessing ecosystem service relationships to ensure effective management strategies are implemented.

4.2 INTRODUCTION

As urban populations continue to grow, there is increasing demand for ecosystem services from urban greenspaces (Haaland and van den Bosch 2015; Niemelä et al. 2010; Tzoulas et al. 2007; United Nations 2015). In response, there are calls to implement management actions that increase the provision of multiple ecosystem services simultaneously within urban greenspaces (McPhearson et al. 2015). However, complex positive and negative relationships exists among ecosystem services that can make this difficult and must be understood if appropriate management choices are to be made (Gaston et al. 2013). This requires quantification of trade-offs and synergies among services under alternative management scenarios that can then be used to choose strategies that result in multifunctional urban green spaces. Yet, the trade-offs and synergies that result from management activities are rarely well understood.

Urban greenspaces provide a multitude of ecosystem services that are the mental and physical benefits humans obtain from ecosystems (Bolund and Hunhammar 1999). In urban systems these include regulating services, such as carbon storage; provisioning services such as food production; and cultural services, such as providing opportunities for exercise (Haase et al. 2014). However, the provisioning of these services are not independent of one another. Relationships exist in the form of trade-offs, where an increase in one service can lead to a decrease in another services, or as synergies, where changing the provisioning of one service leads to a change, in the same direction, of another service (Rodríguez et al. 2006). These relationships arise in response to exogenous or endogenous changes to the system that affect the provisioning of one, or multiple, services referred to as drivers (Bennett et al. 2009). These drivers can include management actions, policy instruments, and natural environmental variability, among others. In urban systems these ecosystem service relationships can be important factors determining whether urban planning provides multifunctional outcomes. For example, Richards and Friess (2017) showed that the development of land for urbanisation in Singapore decreases the provisioning of both carbon storage and opportunities for recreation. in this case a synergistic realetionship leads to a loss in sustainability with respect to both carbon storage and recreation.

Importantly, different management or planning actions can result in different relationships between the same ecosystem services. For example, Zheng et al. (2016) found that a policy promoting farmland expansion in China through clearing forest led to a trade-off between agricultural production and sediment retention, due to the loss of riparian vegetation. However, a policy promoting the development of riparian tree buffers to improve water purification instead led to a synergy between agricultural production and sediment retention (Zheng et al. 2016). Different drivers can also change the magnitude or strength of ecosystem service trade-offs or synergies. For example, to increase crop productivity within agricultural systems, conventional tillage drives a trade-off between water quality regulation and food production, however the use of conservation tillage results in a reduced trade-off between these services as nutrient run-off and soil erosion is reduced (Edgell et al. 2015). These types of variations in trade-offs and synergies are likely to apply equally in urban landscapes, but we understand little about how urban greenspace management may influence the relationships among different services.

Multiple ecosystem services can be synergistic, increasing or decreasing simultaneously under alternative management scenarios (Raudsepp-Hearne et al. 2010; Spake et al. 2017). In urban landscapes, previous studies have found that this is often the case for cultural and regulating services (Rall et al. 2017; Renard et al. 2015). For example, common management actions to increase ecosystem service provisioning in urban greenspaces usually involve revegetation which has previously been found to have a positive link to a number of urban ecosystem services, such a carbon storage and air temperature regulation (Elmqvist et al. 2015; Livesley et al. 2016). However, urban revegetation can either decrease or increase the provisioning of a variety of cultural ecosystem services, depending on the type of revegetation implemented (Shanahan et al. 2015). Therefore, identifying the trade-offs and synergies occurring under different urban revegetation management strategies commonly applied to increase ecosystem service provisioning should help identify which strategies will lead to increases in multiple ecosystem services, rather than an increase in a single ecosystem service. This will allow for more effective management of multiple urban ecosystem services.

In this chapter, I aim to identify whether the trade-offs and synergies occurring among a group of urban ecosystem service change under different urban revegetation management actions. I use the urban greenspace network of Brisbane, Australia, as a case study, assessing five ecosystem services commonly provided in urban greenspaces: opportunities for recreation; opportunities for social interactions; opportunities for relaxation; opportunities for nature interactions; and carbon storage. I first predict the provisioning of each ecosystem service within each park under current conditions using a model that relates park characteristics to the use of the park for the four cultural ecosystem services and carbon storage. To identify the trade-offs and synergies currently occurring between the services, I then conduct a spatial correlation analysis to identify the spatial trade-offs and synergies between the ecosystem services. To determine if these trade-offs and synergies changed under the alternative vegetation management strategies, I then use the ecosystem service models to generate scenarios of alternative revegetation management and calculate the synergies and trade-offs between services for each of these scenarios. I then compare the results of the scenario analysis and spatial correlation analysis to identify whether the trade-offs and synergies among the ecosystem service change under different management.

4.3 METHODS

4.3.1 Study Area

My study area was the Brisbane Local Governmental Area (LGA) in Queensland, Australia (see Figure 3.1). This area includes the city of Brisbane and the surrounding suburbs, occupying 1,380 km^2 and supporting an estimated population of 1.1 million people, as of 2016 (ABS 2016b). This is a rapidly growing region with an expected 200,000 additional residents by 2031 (Queensland Government 2015). Therefore, the capability of Brisbane's greenspaces to provide multiple ecosystem services is critical. Currently, the Brisbane LGA consists of a wide network of public greenspaces, from here on referred to as parks, with a total of 2,872 parks (20,935 ha) used in this study (Figure 3.1). The study area includes one National Park (D'Aguilar National Park), which was excluded from this study because its main function is nature conservation and there is limited public access to much of the park. Therefore, its function is quite different to most other parks in Brisbane.

4.3.2 Quantifying ecosystem services

Carbon storage

A linear regression model to calculate carbon storage across Brisbane's parks under both current and scenario conditions was developed using vegetation structure predictor variables based on vegetation density and vertical strata, as used in Mitchell et al. (2018). Data for the predictor variables were sourced from Mitchell et al. (2018), who used remotely sensed vertical vegetation structure variables derived from Light Detection and Ranging (LiDAR) data (Caynes et al. 2016). The response data consisted of field data of vegetation structure and aboveground carbon in 219 plots across Brisbane that varied in both tree cover and landscape fragmentation, as these are important drivers of vertical vegetation structure. This field data was used to calculate carbon storage for each plot using published allometric equations (Mitchell et al. 2018). To determine which indices to use as predictor variables in my linear regression model, variance inflation factors were calculated for each predictor variable using the "car" package in R (Fox and Weisberg 2011). Vegetation strata indices that returned a variance inflation factor of 5 or larger were regarded as highly correlated and removed from further analyses. Using the carbon storage values calculated from the field plots across Brisbane as the response variable, and the values of the vegetation structure variables at these sites as the predictor variables, I identified the most parsimonious linear regression model. This was done with automatic model selection, using the glmulti package in the statistical program R. The glmulti package calculates the AIC values of all the possible combinations of the predictor variables and selects the model with the lowest AIC value as the best fit model (Calcagno and de Mazancourt 2010). Using this approach, 200 models were tested and the most parsimonious model included the following predictor variables: density of vegetation between 5-10m, density of vegetation above 10m, vertically dense canopy of high trees, and presence of vegetation between 1-5m (AIC difference = 2.4, R^2 = 0.5279) (Table 4.1). This linear regression model was then used to estimate carbon stocks within each park across Brisbane, based on the vegetation structure of the park.

Cultural ecosystem services

Calculation of the provision of the four cultural ecosystem services (opportunities for exercise, opportunities for nature interactions, opportunities for relaxation and opportunities for social interactions) used the cultural ecosystem services models developed in *Chapter 3,* based on response data from a PPGIS survey that identified the parks people visit for different activities within the Brisbane LGA (See Appendix A for further details on the survey methodology). These models are zero-inflated Poisson models that model the number of times a park is visited over a two-week period for activities associated with each of the cultural ecosystem services (Table 4.1). Predictor variables for these models include characteristics of the park and of the people visiting them. To calculate the number of park visits for each cultural ecosystem service in each park across Brisbane, I first characterised values for each of the predictor variables within each park (see Appendix E for data sources). Remotely sensed LiDAR data was used to create data on the vegetation characteristics of each park (proportion of tree cover, grass cover and foliage height diversity), and spatial data provided by Brisbane City Council was used to calculate the size, shape and number of facilities within each park. To calculate values for the socio-demographic and distance from home predictor variables, I used census data to map the distribution of people across the Brisbane LGA and their socio-demographic characteristics. These factors were mapped at the resolution of the census Statistical Area Level 1 (SA1) units, of which there are 2,776 in the Brisbane LGA (mean area: 0.42 km^2, mean population: 423 people). For each SA1 I calculated the population size, mean age, dominant gender, dominant education level, and mean income using the Australian Bureau of Statistics (ABS) Table Builder software (ABS 2016c). A fifth socio-demographic predictor variable ("lifecycle") was used in the cultural ecosystem service models developed in *Chapter 3,* which describes the life stage of a person in terms of whether they have children, their age and relationship status. This variable was removed from the models for this analysis as I was unable to calculate this data for each individual SA1 region, and it was found to have a minimal influence on cultural ecosystem service provisioning (see *Chapter 3*). I calculated the distance from the centroid of each SA1 region to the centroid of each park as an approximation of the "distance from home" variable for people living in each SA1. These environmental and average socio-demographic variables were then used to predict the number of visits to each park for exercise, natural interactions, relaxation and social interactions by a typical person in each SA1. Average socio-demographic variables were used in the models, rather than weighting based on the proportion of people in each socio-demographic category, as I was unable to obtain information on the proportion of each SA1 population with each combination of socio-demographic variables. As previous studies on park use in Brisbane have found that only 60% of residents use public parks (Lin et al. 2014), I multiplied my park visitation values by 60% of the number of people living within each SA1 region, to get the aggregate expected number of visits to each park for each cultural ecosystem service

Table 4.1 Models used to calculate the provisioning of each ecosystem service under current conditions, and under each scenario.

Ecosystem service	Model type	Model	Data source
Carbon storage	Linear regression	**Carbon storage (Mg C)** = density of vegetation between 5-10m + density of vegetation above 10m + vertically dense canopy of high trees + presence of mid-storey vegetation -1	Mitchell et al. (2018)
Opportunities for exercise	Zero-inflated Poisson model	**Number of park visits** = distance form home (m) + presence of amenities + presence of children's play equipment + Presence of access facilities + presence of exercise equipment + presence of dog exercise facilities + gender + education + income + age + proportion of grass + proportion of trees + Foliage height diversity + area of park + shape of park	See *Chapter 3* and Appendix E
Opportunities for natural interaction	Zero-inflated Poisson model	**Number of park visits** = distance form home (m) + presence of amenities + presence of children's play equipment + Presence of access facilities + presence of exercise equipment + presence of dog exercise facilities + proportion of grass + proportion of trees + Foliage height diversity + area of park + shape of park	See *Chapter 3* and Appendix E
Opportunities for relaxation	Zero-inflated Poisson model	**Number of park visits** = distance form home (m) + presence of amenities + presence of children's play equipment + Presence of access facilities + presence of exercise equipment + presence of dog exercise facilities + proportion of grass + proportion of trees + Foliage height diversity + area of park + shape of park	See *Chapter 3* and Appendix E
Opportunities for social interactions	Zero-inflated Poisson model	**Number of park visits** = distance form home (m) + presence of amenities + presence of children's play equipment + Presence of access facilities + presence of exercise equipment + presence of dog exercise facilities + gender + education + income + age + proportion of grass + proportion of trees + Foliage height diversity + area of park + shape of park	See *Chapter 3* and Appendix E

4.3.3 Revegetation management actions

I developed three different revegetation management scenarios which reflect common approaches implemented to enhance vegetation within urban greenspaces, as identified in previous studies (Table 4.2). These scenarios were performed to enhance vegetation from 0-30%, in increments of 5%, within each park.

Table 4.2 Scenarios developed based on the common revegetation management actions used in urban parks.

	Aim	Actions	References
Scenario 1	Increase vegetation structure complexity	Increase mid storey vegetation density by 30% (in 5% increments)	Threlfall et al. (2016), Livesley et al. (2016), Aronson et al. (2014), Mills et al. (2017)
Scenario 2	Increase vegetation cover	Increase tree cover by 30% (in 5% increments)	
Scenario 3	Increase vegetation structure complexity and cover	Increase mid storey vegetation density and tree cover by 30% (in 5% increments)	

Scenario 1 involves increasing mid storey vegetation in areas of pre-existing tree cover. This approach is achieved by planting 1-2m high trees and shrubs within areas that already have tree cover, thereby increasing the mid storey vegetative cover and the complexity of the vegetative structure. Increasing mid storey vegetation has been found to increase biodiversity and therefore the opportunity for nature interactions for park visitors (Bjerke et al. 2006). To calculate carbon storage under Scenario 1, the mid storey vegetation of each park was altered by increasing the value of the "presence of mid storey vegetation" predictor variable within the carbon storage model for each park by 30%, in 5% increments. For parks which already had more than 70% cover of mid storey vegetation, I increased this variable in 5% increments until 100% was reached, with no further change beyond this. To calculate the provisioning of each of the cultural ecosystem services within Scenario 1, I modified the foliage height diversity (FHD) predictor variable in each model. The FHD predictor variable is a measure of how evenly vegetation is distributed among the vertical strata. The mean FHD value for each park was taken from Caynes et al. (2016). Caynes et al. (2016) calculated FHD from Lidar-derived measures of vegetation density across vertical strata. For Scenario 1 I assumed an increase in the mid storey vegetation density by 5 – 30% (in 5% increments) and then recalculated the FHD value for each park.

In Scenario 2, tree cover is increased while maintaining the current complexity in the vegetation structure. This involves planting trees in areas where there is currently no tree cover. This type of revegetation has been found to increase carbon storage, but simultaneously maintaining the “openness” of the understory can also increase cultural ecosystem services, such as social interactions and exercise (Livesley et al. 2016; Shanahan et al. 2015). To calculate carbon storage under this scenario I calculated the average carbon stored in 25m^2 cells in each park in treed areas, and also determined the number of non-treed cells (excluding waterbodies). I then added trees to each of these cells to increase the proportion of tree cover within each park by 30% (in 5% increments) and calculated the additional carbon stored in these cells by multiplying the number of cells by the average amount of carbon (Mg C) stored in a single cell within that park. This allowed me to estimate the additional carbon stored if tree cover was increased from 5 – 30% without any changes to the vegetation complexity within the park. For parks where tree cover could not be increased by 30%, because more than 70% of the park is currently treed, I only added tree cover until 100% tree cover was reached. To calculate the provisioning of the cultural ecosystem services the proportion of tree cover in each park was increased by 5 – 30% (in 5% increments). I capped the total amount of tree cover possible at 100%. The cultural ecosystem service models then calculated new values for each cultural service in each park.

In Scenario 3, both tree cover and the presence of mid-storey vegetation is increased within each urban park. This involves combining the approaches for Scenarios 1 and 2. This management action has previously been found to increase carbon storage and nature interactions (Bjerke et al. 2006; Livesley et al. 2016). To calculate carbon stored in Scenario 3, I altered both the “presence of mid storey vegetation” and increased amount of tree cover within each park by combining the methods used for scenarios 1 and 2. To calculate the cultural ecosystem services, both the proportion of tree cover and the density of mid storey vegetation were increased by 5 – 30% (in 5% increments). New predictions for each of the cultural ecosystem services for each park were then made.

4.3.4 Analysing ecosystem service relationships

To compare the trade-offs and synergies occurring spatially and temporally between the ecosystem services, I used two approaches. I first identified trade-offs and synergies spatially using a spatial correlation approach which calculates correlations coefficients between each pair of ecosystem services to determine whether their spatial occurrences are negatively or positively correlated (Mouchet et al. 2014). I then identified trade-offs and synergies temporally under each management action using a scenario analysis, in which the provisioning of each pair of ecosystem services is compared at different points in time, under the different management scenarios (Mouchet et al.

2014). Using this approach it is possible to identify trade-offs and synergies by identifying which pairs of services increase or decrease together over time, and under each scenario.

For the spatial-correlation approach, I performed a correlation analysis of the ecosystem services occurring within the parks under the current scenario using Spearman's Rank Correlation Coefficients. As larger parks are capable of providing larger amounts of each service, I standardised the ecosystem service values for each park by dividing them by the area of the park. Ecosystem services whose provision was spatially positively correlated and statistically significant ($R^2 > 0.7$, $p \leq 0.05$) were regarded as having a synergistic relationship, and services that were negatively correlated ($R^2 < -0.7$, $p \leq 0.05$) were regarded as trade-offs.

For the scenario-based approach to quantifying trade-offs and synergies, the relationships between the provisioning of each ecosystem service predicted under each scenario were plotted. To ensure comparability with the spatial correlation approach, ecosystem service values were divided by area of the park (ha) and I then plotted the average amount of each ecosystem service provided per ha under each scenario.

I then determined whether the trade-offs and synergies were changing over time under the different management actions. This was done by comparing the trade-offs and synergies occurring between the ecosystem services spatially, which were identified using the spatial correlation approach, to the relationships occurring over time under the different management actions, which were identified using the scenario analysis.

4.4 RESULTS

Under current conditions, the ecosystem service models estimate that Brisbane parks provide 1.87 million Mg of aboveground carbon storage, 1.016 million park visits over a two week period for exercise, 395,048 park visits over a two week period for nature interactions, 313,280 park visits over a two week period for relaxation, and 1.421 million park visits over a two week period for social interactions (Figure 4.1). The spatial correlation analysis identified significant synergies between all the cultural ecosystem services (opportunities for exercise, nature interactions, relaxation and social interactions) (Figure 4.2). A synergy was also recorded between nature interactions and carbon storage. Significant trade-offs were recorded between carbon storage and social interactions, and between carbon storage and relaxation.

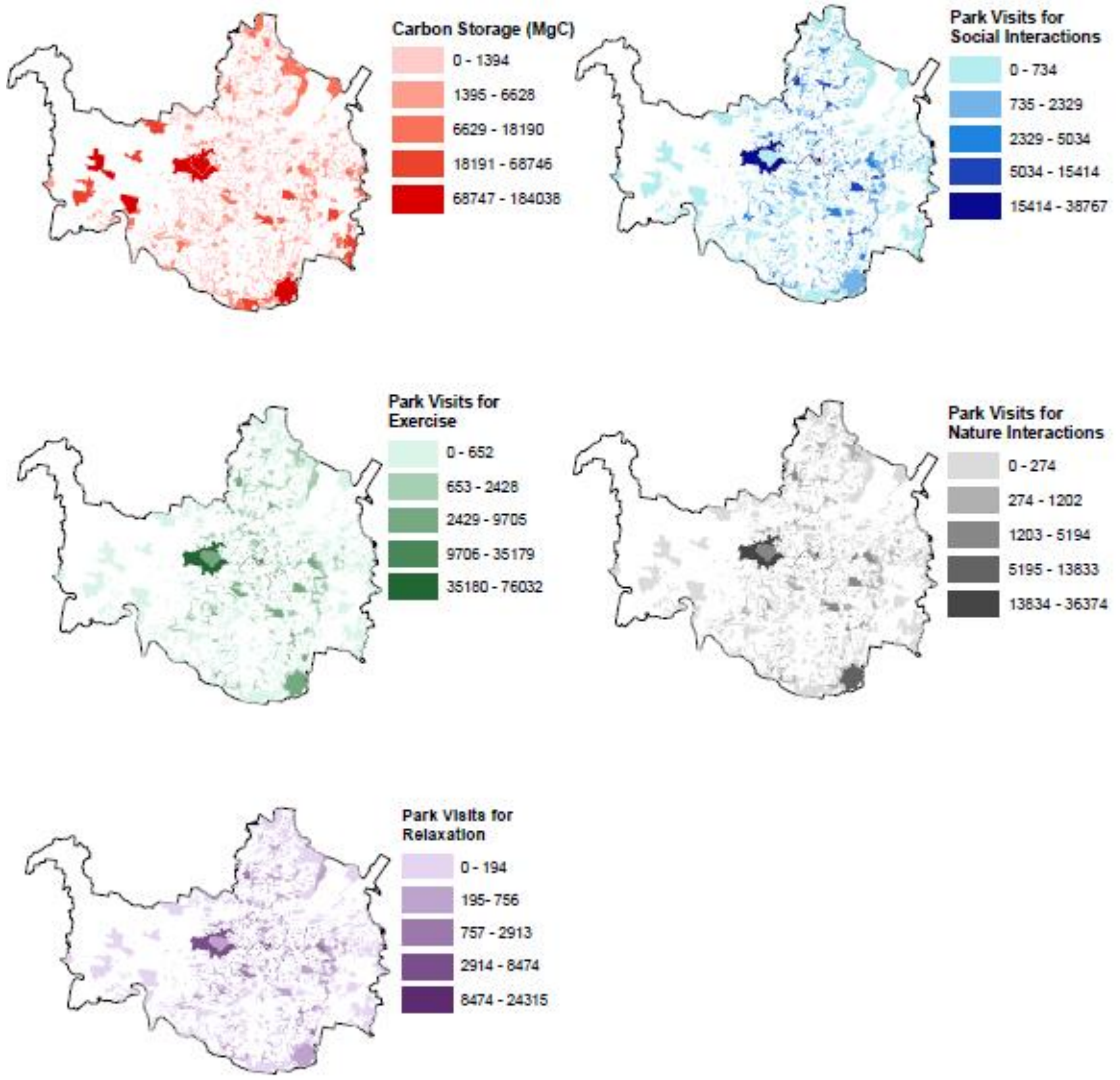

Figure 4.1 Spatial distribution of each ecosystem service within Brisbane LGA's park network under current conditions. Values represent total values of ecosystem service provision for each park.

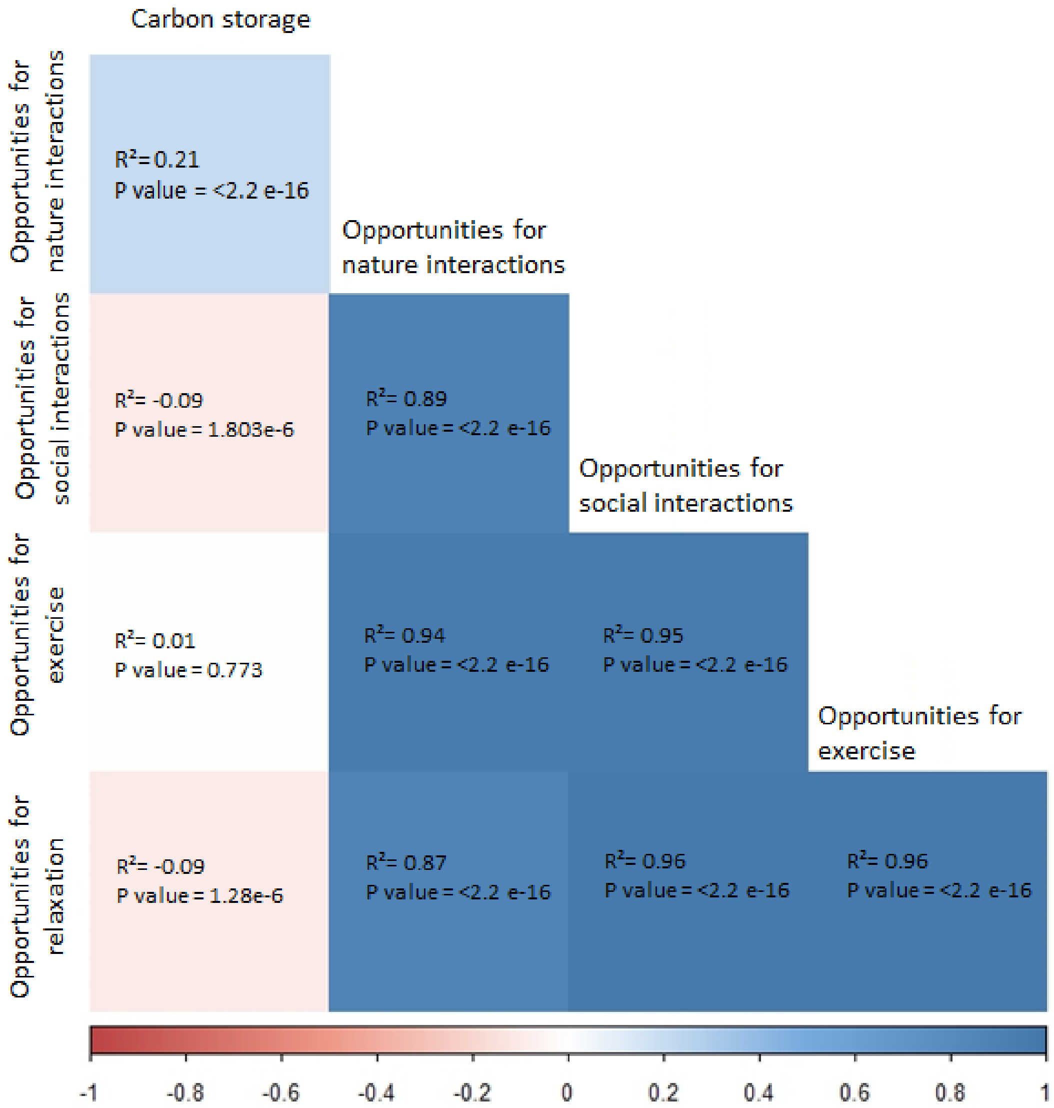

Figure 4.2 Spatial correlations between ecosystem services, based on current conditions, using Spearman's Rank Correlation Coefficients, with the colour gradient representing the R^2 value. $R^2 > 0.7$ indicates the relationship between two services is positive, $R^2 < 0.7$ indicates the relationship between two services is negative, and $p \leq 0.05$ indicates whether this relationship is significant.

4.4.1 Scenario 1

There was very little change in the provisioning of the five ecosystem services as mid storey vegetation density increased (Figure 4.3). When density of mid storey vegetation was increased by 30%, the number of park visits for relaxation increased by only 0.07% across Brisbane and total carbon stocks within the parks increased by only 1.22% (an increase of 25,524 Mg C). The total number of park visits for exercise, nature interactions and social interactions decreased by less than 1% (Exercise = -0.29%, Nature Interactions = -0.27%, social interactions = -0.10%). There was a slight synergistic relationship between carbon storage and relaxation, and a slight synergistic relationship between exercise, nature interactions and social interactions (Figure 4.4). When the ecosystem services were plotted against one another, the slope of the line was slightly negative between carbon storage and exercise, nature interactions and social interactions, as well as between relaxation and exercise. This suggests minor trade-offs occurring between these services.

4.4.2 Scenario 2

Under this scenario, carbon storage and park visits for nature interactions increased as tree cover increased (Figure 4.3). At a 30% increase in tree cover, the carbon stocks within the parks increased by 29.63% (873,227 Mg C), and the total number of park visits for nature interactions increased by 8.03% (34,486 visits). However, there was a negative relationship between tree cover and the provisioning of exercise, relaxation and social interactions. At a 30% increase in tree cover, the total number of park visits for exercise decreased by 5.17% (49,960 visits), relaxation decreased by 17.62% (46,965 visits), and social interactions decreased by 20.85% (24,5295 visits). There was a synergistic relationship between carbon storage and nature interactions, and between exercise, relaxation and social interactions (Figure 4.4). Trade-offs were observed between carbon storage versus exercise, relaxation and social interactions, as well as between nature interactions versus exercise, relaxation and social interactions.

4.4.3 Scenario 3

Under this scenario, carbon storage and park visits for nature interactions increased as both tree cover and mid storey vegetation density increased (Figure 4.3). With a 30% increase in both tree cover and mid storey vegetation density, the average carbon stocks within a park increased by 27.45% (784,728 Mg C) and the total number of park visits for nature interactions increased by 7.76% (33,247 visits). However, the provisioning of exercise, relaxation and social interactions decreased on average as tree cover and mid storey vegetation density increased. At a 30% increase in tree cover, the average number of park visits for exercise decreased 5.47% (52,712 visits), relaxation decreased by 17.54% (46,777 visits), and social interactions decreased by 20.97% (246,440 visits). Therefore, under this vegetation management scenario, synergies were observed between carbon storage and nature interactions, as well as between exercise, relaxation and social interactions (Figure 4.4). Trade-offs were observed between carbon storage and exercise, relaxation and social interactions, as well as between nature interactions and exercise, and relaxation and social interactions.

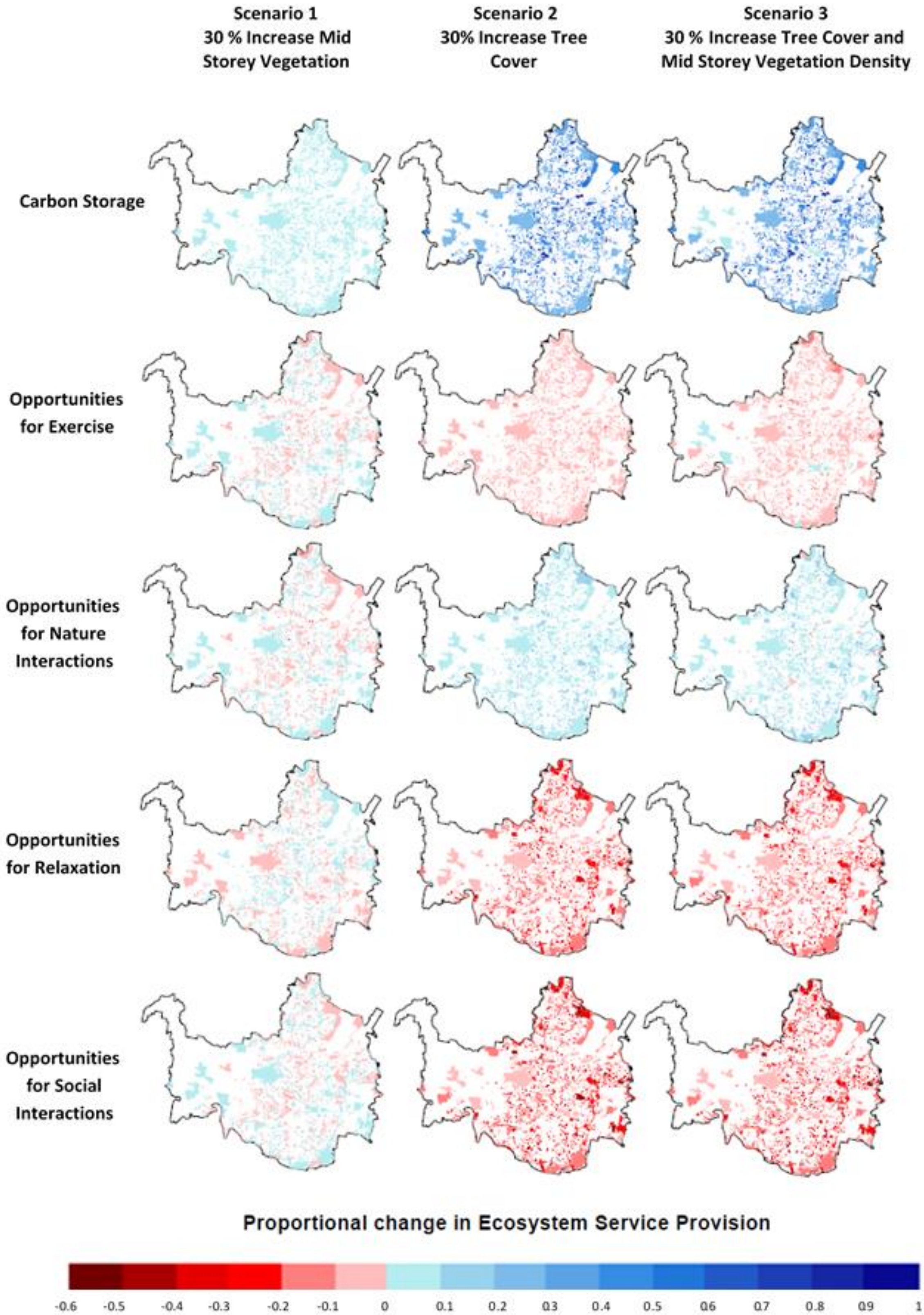

Figure 4.3 Proportional change in the provision of each ecosystem service under each scenario. Changes are relative to ecosystem service provision under current conditions.

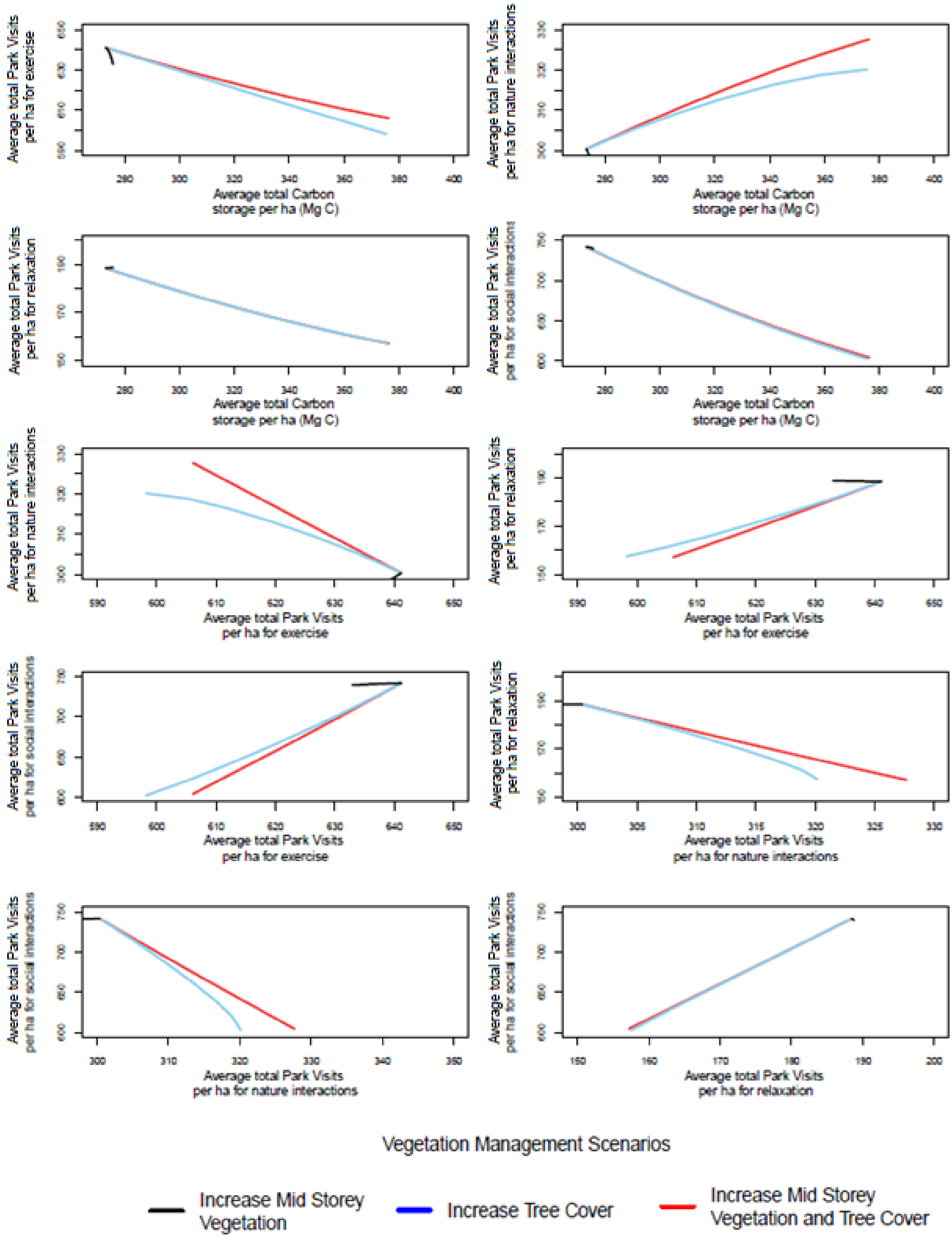

Figure 4.4 Plots depicting the average provisioning of each pair of ecosystem services under each scenario, as mid storey vegetation density and/or tree cover are increased in 5% increments (to a maximum increase of 30%). Negative trends indicate trade-offs, and positive trends indicate synergies.

4.5 DISCUSSION

To identify effective management strategies of multiple ecosystem services within urban landscapes it is necessary to first understand the trade-offs and synergies occurring between the ecosystem services under alternative management choices (Gaston et al. 2013). In this study I found that revegetation management actions that focus on changing different vegetation variables can generate different relationships between the carbon storage and four different cultural ecosystem services. Only some of these relationships were captured by the spatial correlation approach, demonstrating the importance of incorporating drivers into the assessments of ecosystem service relationships to implement effective management that avoid trade-offs occurring between services. These results support the findings of previous studies (Briner et al. 2013; Kain et al. 2016; Kremen 2005; Turkelboom et al. 2018) that found that drivers of ecosystem service relationships play a key role in the effectiveness of management actions and policy instruments to target multiple ecosystem services.

Revegetation actions focusing on increasing mid storey vegetation had little impact on the provisioning of any of the ecosystem services included in my study, resulting in only minor relationships between the ecosystem services. It is unsurprising that there was little effect on carbon storage with this scenario, as mid storey vegetation stores much less carbon than trees as it usually consists of less woody biomass (Davies et al. 2011). However, these results also suggest that increasing mid storey vegetation has little impact on the cultural ecosystem services as well. In a study on the role of vegetation density on park use for recreation, Bjerke et al. (2006) found that socio-demographic variables, rather than the type of activity more strongly influenced a person’s decision to visits parks with varying levels of vegetation density. This suggests that rather than affecting the type of cultural ecosystem services a park is used for, increasing mid storey vegetation may in fact affect the type of people visiting the park for the same cultural ecosystem services. This dynamic was not captured by my analysis. Under the scenarios focusing on increasing tree cover, trade-offs and synergies occurred leading to positive and negative changes in the provisioning of multiple ecosystem services. Tree cover has previously been found to have a negative relationship with park activities related to exercise, relaxation and social interactions (Adinolfi et al. 2014; Shanahan et al. 2015), and a positive relationship with nature interactions (Soga and Gaston 2016), matching the patterns of trade-offs and synergies identified in this study. Furthermore, carbon storage has a direct positive link to tree cover, making tree cover a strong driver of carbon storage (Nowak et al. 2013). This leads to a synergy occurring with nature interactions as higher urban tree cover can increase species diversity which makes for stronger interactions between people and nature (Beninde et al. 2015).

The synergies and trade-offs I identified between ecosystem services differed depending on the methods used to identify them. Spatial correlation only identified significant synergistic relationships between carbon storage and opportunities for relaxation and carbon storage and social interactions. However, the scenario analysis indicated that trade-offs and synergies occur between the cultural ecosystem services as well. It is possible that the spatial correlation approach was unable to capture the relationships between the cultural ecosystem services as it was simply confounding between a large number of different environmental and social drivers that affect the provisioning of the cultural services (Wilkerson et al. 2018). However, carbon storage relies solely on the environmental variables allowing for a more accurate interpretation of the carbon storage relationships. Therefore, the ability for the scenario analysis to account for these variables, and incorporate drivers into the assessments of trade-offs and synergies, allows for a better understanding of how drivers influence ecosystem service trade-offs and synergies, and the consequences for effective management.

This study highlights that implementing actions to effectively manage multiple urban ecosystem services, should rely on assessments of ecosystem service trade-offs and synergies that explicitly consider drivers. Approaches that assume ecosystem service relationships are fixed, and do not depend on drivers, such as the spatial correlation approach, cannot accurately determine how management actions will affect these relationships. Currently, when assessing trade-offs and synergies to identify suitable management options, decision-makers often use spatial correlations, such as overlap analysis (Lee and Lautenbach 2016; *Chapter 2*). This can result in decisions to introduce policy instruments and management actions that lead to perverse outcomes and unexpected declines in multiple ecosystem services (Briner et al. 2013). On the other hand, methods that explicitly incorporate drivers into the assessment, such as a scenario analysis, are able to better identify how these relationships are likely to vary over space and time (Birkhofer et al. 2015). This is particularly important for urban planners, as knowledge of the drivers that underpin trade-offs and synergies among many urban ecosystem services is still limited (McPhearson et al. 2015). Using a scenario analysis that explicitly considers drivers when identifying ecosystem service management actions in urban regions will assist in both increasing our understanding, and ensuring effective management of urban ecosystem services and multifunctional urban greenspaces.

This chapter focuses on the ecosystem service trade-offs and synergies that arise under different revegetation management scenarios. As vegetation differs significantly across different countries and regions (Foley et al. 2005), it is likely that that these relationships may be different in different cities. For example, the relationships between vegetation structure and carbon storage could differ significantly between tropical, subtropical and temperate urban zones (Dobbs et al. 2014). Furthermore, socio-demographic characteristics and cultural values differ across urban areas and affect the ecosystem services people receive from urban greenspaces, which is likely to impact ecosystem service relationships (Wilkerson et al. 2018). This analysis only compared the ecosystem service trade-offs and synergies identified using two methods, one that explicitly considered drivers and one that did not. It is possible that other methods might be better able to quantify the relationships occurring, depending on how drivers are considered in each assessment. For example, field experiments, regarded as the gold standard of mechanistic approaches, might identify more accurate assessments of trade-offs and synergies as this method allows for the isolation of the mechanisms of specific drivers, which is not possible with either the spatial correlation or scenario analysis approaches (Mouchet et al. 2014). I recommend that further analyses are performed to compare different methods that identify trade-offs and synergies and evaluate these against experimental data. This will provide further clarity about the methods that can best identify relationships between ecosystem services under alternating management actions.

4.6 CONCLUSION

There is increasing pressure for urban greenspaces to be multifunctional and capable of providing a wide variety of benefits to human wellbeing. However, different management actions can generate different trade-offs and synergies among the same ecosystem services. Methods that explicitly consider the drivers underpinning ecosystem service provisioning can improve our ability to identify how different management actions drive ecosystem service trade-offs and synergies. This highlights the importance of using methods that incorporate drivers to identify trade-offs and synergies. Furthermore, it contributes to our growing understanding of the nature of the drivers that underpin ecosystem service trade-offs and synergies, urban landscapes.

CHAPTER 5

MANAGEMENT PRIORITIES FOR EQUITABLE DISTRIBUTION OF URBAN ECOSYSTEM SERVICES.

5.1 ABSTRACT

To ensure the mental and physical wellbeing of growing urban populations, urban greenspaces must be managed to increase the provisioning of multiple ecosystem services. This often requires identifying which management actions to implement, and where, that will increase the provisioning of multiple services and minimise costs associated with management, while ensuring social equity in the distribution of ecosystem service increases across the landscape. Unfortunately, no studies have identified the optimal spatial allocation of multiple actions across an urban landscape to achieve increases in the provisioning of multiple ecosystem services, while also minimising costs and ensuring social equity. Using the urban park network of Brisbane, Australia, I identified the optimal arrangement of management actions to increase the provisioning of carbon storage and a combination of cultural ecosystem services, and assessed the impact that accounting for social equity has on both the amount these services can be increased by and the associated management costs. Using conservation spatial planning software, I determined the specific management actions to allocate to each park to achieve targeted increases in each ecosystem service, under both socially equitable and socially inequitable scenarios. My results show that increasing the provisioning of the ecosystem services to different amounts affects the spatial arrangement of management actions. Furthermore, ensuring social equity reduced the amount each ecosystem service could be increased by, and increased average management costs by 78%. If implemented for urban park policies, my novel approach could help inform the design and management of urban parks to achieve specific targets in multiple ecosystem services to improve the wellbeing of all urban residents.

5.2 INTRODUCTION

There is increasing pressure to manage urban landscapes in ways that ensure they can provide multiple ecosystem services and contribute to human wellbeing (Bennett et al. 2015; Stott et al. 2015; McPhearson et al. 2015). Managing multiple ecosystem services across urban landscapes is difficult as different management actions will affect each ecosystem service differently (Andersson et al. 2014; Gaston et al. 2013). Furthermore, social equity must be considered when increasing the provisioning of ecosystem services across the landscape to ensure urban residents have equal access to the ecosystem services they require (Wilkerson et al. 2018), and that no one area is burdened with all of the management costs of providing ecosystem services (Pascual et al. 2017). Finally, as decision makers are often faced with strict budgets, it is important that management actions are implemented at minimum cost. How to achieve environmental benefits alongside social equity and economic benefits is therefore a key challenge that decision makers face when managing multiple urban ecosystem services (Halpern et al. 2013; Wu 2013).

Urban areas provide a wide array of ecosystem services (Bolund and Hunhammar 1999). To increase the provisioning of ecosystem services within urban parks, managers and research have traditionally focused on actions that increase the provisioning of individual ecosystem services (Gaston et al. 2013). However, management actions influence the provisioning of multiple ecosystem services, and introducing management actions to increase the provisioning of one service could lead to decreases in the provisioning of other services, or lower levels of increases in other services (Bennett et al. 2009). For example, to increase the provisioning of cultural ecosystem services within parks, increased infrastructure for access and use, such as roads and facilities, could be implemented (Rigolon 2016). This is likely to require impacts such as the removal of tree cover that can reduce the park's capacity to provide carbon storage and air temperature regulation (Lauf et al. 2014). It can therefore be difficult to implement management actions that increase the provisioning of multiple services. For this reason, urban planners and decision makers are moving towards focusing on implementing multiple management actions within public urban greenspaces, hereon referred to as parks, and targeting the provisioning of multiple groups of ecosystem services (see Brisbane City Council (2013)).

In managing ecosystem service provision across urban landscapes, it is important to consider social equity, in both access to the ecosystem services, as well as shared responsibility in managing the services when there are multiple local councils present that are responsible for separate parts of the landscape (Jennings et al. 2017; Jennings et al. 2016). Access to urban ecosystem services is often highly unequal and influenced by socio-economic status (Jenerette et al. 2011; Dobbs et al. 2014a; Jennings et al. 2016; Rigolon 2016). For example, in a study of the spatial distribution of multiple ecosystem services across the city of Melbourne, Dobbs et al. (2014a) found that people living within highly affluent suburbs had better access to a wide range of regulating and cultural services due to the design, level of maintenance of parks and tree cover within these suburbs. This inequality in access could lead to serious consequences to the mental and physical wellbeing of a large number of urban residents (Jenerette et al. 2011). Furthermore, a situation where one or a few parks within an urban landscape provide the majority of an ecosystem service can reduce social equity, as well as result in unequal financial burdens in managing these ecosystem services (Schröter et al. 2017; Pascual et al. 2017). Therefore, management actions to increase multiple ecosystem services should be strategically conducted across urban landscapes to ensure social equity, so that urban residents have equal access to different ecosystem services and the responsibility for managing them.

Identifying where to allocate different management actions across an urban landscape to ensure that targeted increases in multiple ecosystem services are achieved, while accounting for social equity and cost constraints, is a complex problem. However, these types of problems can be solved using structure decision making and optimisation techniques. Optimisation has been accepted as an important decision-making tool for prioritising actions across landscapes for natural resource management, including the management of ecosystem services (Chan et al. 2006; Kukkala and Moilanen 2012; Law et al. 2016). The approach focuses on solving resource allocation problems that involve prioritising actions or resources to achieve a target performance, such that cost is minimised (Kukkala and Moilanen 2012). For the management of multiple ecosystem services, optimisation requires (i) identifying the potential actions that increase the provisioning of these services and by how much, and (ii) determining constraints, such as the amount to increase each ecosystem service to. There are a number of decision support tools that can then be used to solve the optimisation problem, including tools designed primarily for optimal spatial zoning, such as Marxan (Chan et al. 2006) and Zonation (Snäll et al. 2016), as well as more general optimisation tools, such as integer linear programming (Beyer et al. 2016). A number of studies have previously focused on identifying where to implement management actions to spatially optimise ecosystem service provisioning in non-urban landscapes (Bryan et al. 2010; Law et al. 2016; Luck et al. 2012). However, I am not aware of any studies that have focused on the optimal arrangement of multiple management actions to increase the provisioning of multiple ecosystem services within urban landscapes. Further, being able to solve these decision making problems while also achieving social equity would be a major advance for urban sustainability and to help ensure the mental and physical wellbeing benefits of residents (Niemelä et al. 2010).

In this paper, I use optimisation to identify the optimal spatial allocation of management actions to increase the provision of multiple ecosystem services. I apply this approach to the management of the park network in Brisbane, Australia and evaluate the impacts of accounting for social equity on priorities and costs. I focus on increasing the provision of two ecosystem services that are commonly considered in urban park management plans and where social equity is regarded as important for either management or access: carbon storage, and the sum of four cultural ecosystem services (opportunities for exercise, nature interactions, relaxation and social interactions), hereon referred to as cultural ecosystem services. For carbon storage, social equity is important in ensuring an even distribution of the responsibility of managing the trees that store carbon across the landscape (McDermott et al. 2013; Pascual et al. 2017). For the cultural ecosystem services, social equity is important in ensuring equality of access so that all urban residents can easily access and receive these ecosystem services (Rigolon 2016). I show that by applying spatial optimisation methods it is possible to explicitly identify where to implement different management actions to achieve specified increases in multiple ecosystem services at minimum cost. This advances our understanding and ability to create sustainable landscapes that can provide multiple ecosystem services.

5.3 METHODS

5.3.1 Study region

The Brisbane Local Governmental Area (LGA) is a subtropical region located in Queensland, Australia (Figure 5.1). There are a wide range of public urban greenspaces within the Brisbane LGA, including conservation reserves, community parks and sporting fields, with the facilities and vegetation characteristics varying across these parks (Brisbane City Council 2006). To ensure there is equality in the benefits people receive from increases in ecosystem service provisioning, there is a need to strategically allocate park management actions across Brisbane to evenly distribute improvements in ecosystem service provision and ensure greater social equity in both access and management responsibilities. The study area includes one National Park (D'Aguilar National Park), which was excluded from the study because its main function is nature conservation and there is limited public access to much of the park. Therefore, its function is quite different from most other parks in Brisbane.

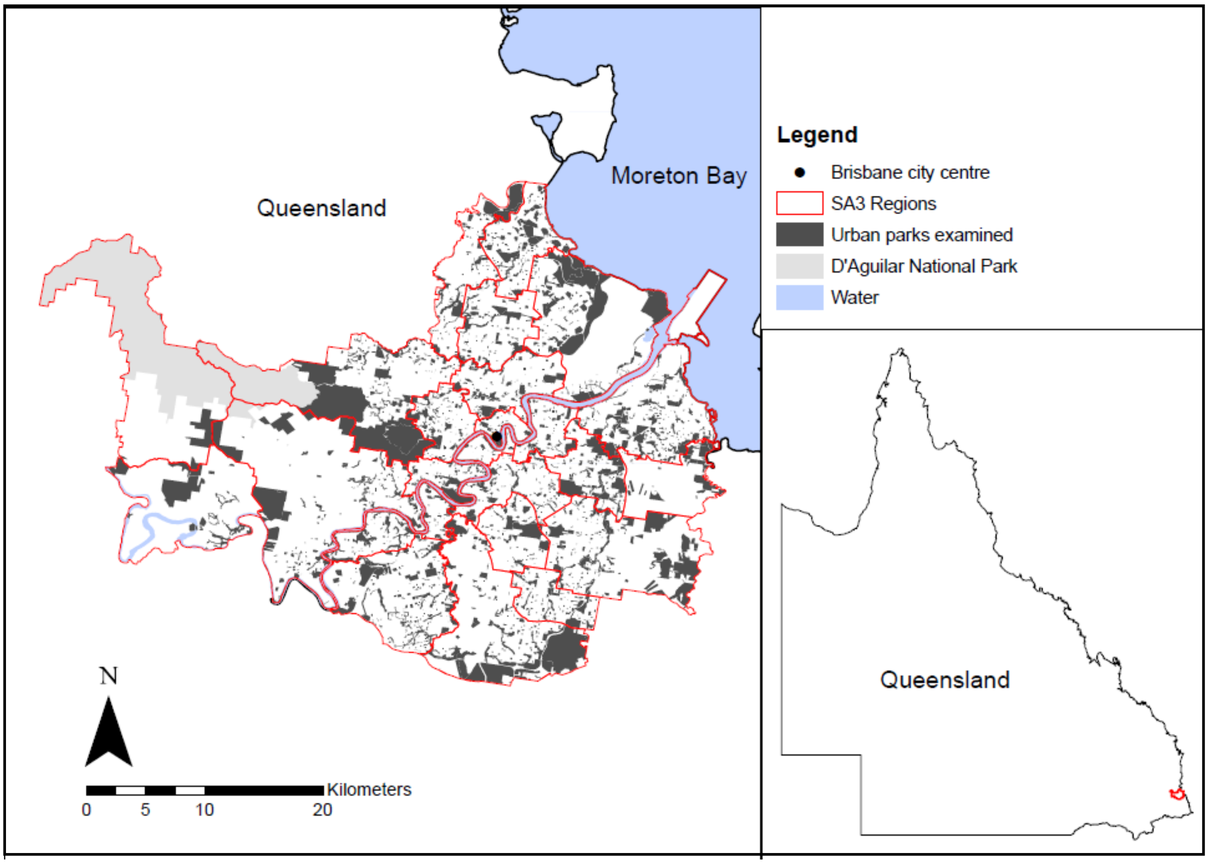

Figure 5.1 Brisbane Local Governmental Area, including the location of the urban parks assessed in this study, and the SA3 regions that this area is divided into for the social equity management scenario.

5.3.2 Decision problem and optimisation approach

Our goal was to identify where to allocate different management actions across Brisbane's park network to achieve specific targets in the provisioning of carbon storage and cultural ecosystem services at minimum cost, when social equity is accounted for and when it is not. First, I identified a list of management actions that could be implemented to increase the provisioning of one, or both, ecosystem services. Secondly, I calculated the provisioning of each ecosystem service under each management action, within each park. Thirdly, I calculated the costs to implement each management actions within each park. Finally, I identified the specific targets that I wanted to increase each ecosystem service to. To then solve my decision problem, and find the optimal arrangement of management actions, I imported the information on the management actions and targets into an optimisation decision support tool. To determine the environmental and economic costs of accounting for social equity in the management of multiple urban ecosystem services, I compared the optimal management solutions and costs for a range of ecosystem service targets under two different management scenarios: one where social equity in the ecosystem services was accounted for, and one where social equity in the ecosystem services was not accounted for.

5.3.3 Management actions and associated changes in the ecosystem services

We identified a list of feasible management actions that can be implemented within each park to influence the provisioning of the focal ecosystem services (Table 5.1). These management actions reflect common improvements that are adopted within urban parks (McCormack et al. 2010). Management Action 1 involves replacing 80% of grass cover with tree cover within a given park. Tree cover directly increases carbon storage (Davies et al. 2011) and can have either negative or positive influences on cultural ecosystem services (see *Chapter 3*). Due to the design and multi-functional use of most urban parks, it may therefore not be the best option to increase area of tree cover to 100% (Shanahan et al. 2015). I therefore limited tree cover increases to 80%, while maintaining the same level of vegetation structural complexity as the original tree cover. Management Action 2 involves replacing 80% of tree cover with grass cover within a given park, as grass cover has previously been found to have a positive effect on some cultural services, but a negative effect on carbon storage (Davies et al. 2011; McCormack et al. 2010). Management Actions 3, 4 and 5 include adding facilities to parks where they are currently not present. Facilities have been found to have a positive influence on the provisioning of some cultural ecosystem services, with little to no influence on carbon stocks (see *Chapter 3*). Management Action 3 involves adding a children's playground to parks, Management Action 4 adding a footpath the length of the park, and Management Action 5 adding amenities, such as barbeques, toilet blocks and benches. Management Action 6 involves no change to the current condition of a given park, and therefore will have no impact on ecosystem service provision. Management Action 7 is a combination of actions 1, 3, 4 and 5 and involves replacing 80% of grass cover with tree cover while adding facilities within a park. Management Action 8 is a combination of actions 2, 3, 4 and 5 and involves replacing 80% of tree cover with grass cover and adding a combination of facilities to a given park.

Table 5.1 Management actions and their associated costs. Costs are all in AUD$.

	Description	**Associated costs per park**
Management action 1	Replace 80% of grass cover with tree cover over	Add tree cover = $11.26 per m^2 Removing grass = $151.65 per m^2
Management action 2	Replace 80% of tree cover with grass cover over	Add grass = $9.52 per m^2 Remove tree cover = $151.65 per m^2
Management action 3	Add a children's playground to parks with no playgrounds	Medium-sized playground = $46,326.04
Management action 4	Add a footpath shortest distance across the park	Footpath = $209.50 per m
Management action 5	Add a toilet block, barbeque, seating and shade to parks where these are currently absent	Toilet block = $120,346 Electric barbeque = $8,442.18 Bench = $1,197.52 Shade structure = $6,818.50
Management action 6	Do nothing	$0
Management action 7	Replace 80% of grass cover with tree cover, add a children's playground to parks with no playgrounds, add a footpath measuring the width of the park, add a toilet block, barbeque, seating and shade to parks where these are currently absent	Combined costs for Management Actions 1,3,4 and 5
Management action 8	Replace 80% tree cover with grass cover, add children's playground to parks with no playgrounds, add a footpath measuring the width of the park, add a toilet block, barbeque, seating and shade to parks where these are currently absent	Combined costs for Management Actions 2,3,4 and 5

We then used the ecosystem service models previously developed in *Chapter 4* to estimate the provision of carbon storage and the cultural ecosystem services in each park under each management action (see Table 4.1 in *Chapter 4* for the ecosystem service models). This involved changing the current environmental and facility characteristics of each park to reflect their conditions under each management action and then predicting the provision of each service using the regression equations (see Appendix D for a list of data sources for the predictor variables in the models). To calculate a value for the cultural ecosystem services within each park, I calculated the expected number of park visits over a two week period for the four individual cultural services it consists of (opportunities for exercise, nature interactions, relaxation and social interactions) and then summed these four values together for each park to give a single cultural ecosystem service value. As previous studies on park use in Brisbane have found that only 60% of residents use public parks (Lin et al. 2014), I multiplied my park visitation values by 60% to get the aggregate expected number of visits to each park for the cultural ecosystem services.

5.3.4 Management action costs

The costs of implementing each management action was calculated based on cost data obtained from Logan City Council, the neighbouring city council to Brisbane, as I was unable to source this data directly for the Brisbane LGA (Table 5.1) (Logan City Council 2015). These costs included the supply of materials and installation, as well as gardening supplies and earthworks to add and remove grass or tree cover. As cost data was based on prices from 2009, the Australian Bureau of Statistics Inflation Calculator was used to estimate the 2017 costs by indexing them to inflation (ABS 2018). Using the estimated costs for each management action (Table 5.1), I then calculated the cost of implementing each management action within each park. As the features and size of each park differed, the costs associated with each management actions also differed for each park.

5.3.5 Management scenarios

To develop the socially equitable scenario, I used the container method, as described in Lindsey et al. (2004). The container approach defines social equity as the presence of a resource (such as an ecosystem service) within a specified area or zone (Lindsey et al. 2004). Therefore, for my socially equitable scenario I set ecosystem service targets to ensure equal improvements in ecosystem services occurred across designated zones within the Brisbane LGA. I split the Brisbane LGA into zones, using Statistical Area Level 3 (SA3) regions (Figure 5.1), where SA3s are census statistical units designed to cluster suburbs that have similar socioeconomic characteristics, with each containing the same number of people (ABS 2016a). Optimisations were then conducted for each SA3 zone separately. Any SA3 regions not containing any public parks (n = 3) were removed from the analysis. For the socially inequitable scenario, I conducted the spatial optimisation analysis by setting global targets for ecosystem services, with no regard to zone targets. Under this scenario, there were global targets for each ecosystem service, allowing ecosystem services to be increased in any park across the Brisbane LGA, due to the allocation of management actions, to achieve a targeted increase in the ecosystem services.

5.3.6 Ecosystem service targets and optimisation

We used the Prioritizr package in R (Hanson et al. 2018) to identify the optimal spatial arrangement of the management actions across Brisbane's urban parks that meets a targeted increase in the provision of carbon storage and the cultural ecosystem services at minimum cost, under both the socially inequitable and socially equitable scenarios. The Prioritizr package is a systematic conservation planning tool that uses integer linear programming to build and solve a broad range of conservation planning problems under a range of constraints, such as the target amount to increase an ecosystem service by (Hanson et al. 2018). For the socially inequitable scenario, I set a global target to achieve for each ecosystem service, and for the socially equitable scenario I set targets for each individual SA3 region in Prioritizr.

I chose to set targets as percentage increases in ecosystem service provisioning. To determine the range of feasible targets I first solved the optimisation problem while incrementally increasing the carbon storage target by 1%, with a zero increase target for the cultural ecosystem services, until the carbon storage target was infeasible. A zero increase target represents no set target for the ecosystem service. Therefore under a 0% target, management actions can be place that increase, decrease or do not change the value for the cultural ecosystem services. This was then repeated for the cultural services, with the carbon storage target increase set to zero. I then looped through all possible combinations of feasible targets for carbon storage and the cultural ecosystem services in 1% increments and found the optimal solution and cost. For the socially equitable scenario, I repeated the same process but for each individual SA3 region within the Brisbane LGA. This ensured that management actions were spatially allocated to ensure even distribution of increases in both ecosystem services across the Brisbane LGA. For each SA3 and each target I recorded the optimal solution and the cost.

All analyses and programming were conducted in the R statistical package (v.3.4.1) with integer programming within the Prioritizr package solved with Gurobi (v.7.0.1).

5.4 RESULTS

There were 1,407 feasible targets for carbon storage and cultural ecosystem services in the socially inequitable scenario (Figure 5.2). Under this scenario, the most carbon storage could be increased by was 13% (243,414 Mg C), which was achievable when there was a cultural ecosystem service target ranging from 0% to 93% increase. The highest feasible target for the cultural ecosystem services was a 101% increase (an increase of 5,189,237 park visits over a two-week period), but this was only feasible when the carbon storage target was 0%. For the socially equitable scenario, there were 129 feasible targets for carbon storage and the cultural ecosystem service (Figure 5.2). Here, the highest feasible target for carbon storage was a 2% increase within each SA3 region (a total increase of 37,475 Mg C across Brisbane). This was achievable when there was a cultural ecosystem service target ranging from 0% to a 42% increase. The highest feasible target for the cultural ecosystem services was a 42% increase within each SA3 region (a total increase of 1,808,269 park visits over a two-week period). This was achievable when there was a carbon storage target ranging from a 0% to a 2% increase. This indicates that the maximum feasible targets for both ecosystem services cannot be achieved simultaneously under the socially inequitable scenario across Brisbane's urban greenspace (Figure 5.2).

The costs associated with implementing management actions to achieve each ecosystem service target varied across the two scenarios (Figure 5.2). The costs associated with achieving targets under the socially equitable scenario were, on average, 78% higher than the costs associated with the targets under the socially inequitable scenario. For example, a 40% target in the cultural ecosystem services with a 0% target for carbon storage cost $36.98 million under the socially inequitable scenario. However, under the socially equitable scenario the same target cost $62.45 million (Figure 5.2). The costs also varied across the feasible target range for the two scenarios. Under both equity scenarios, the costs for increasing carbon storage targets increased more rapidly than the costs for increasing the cultural ecosystem services.

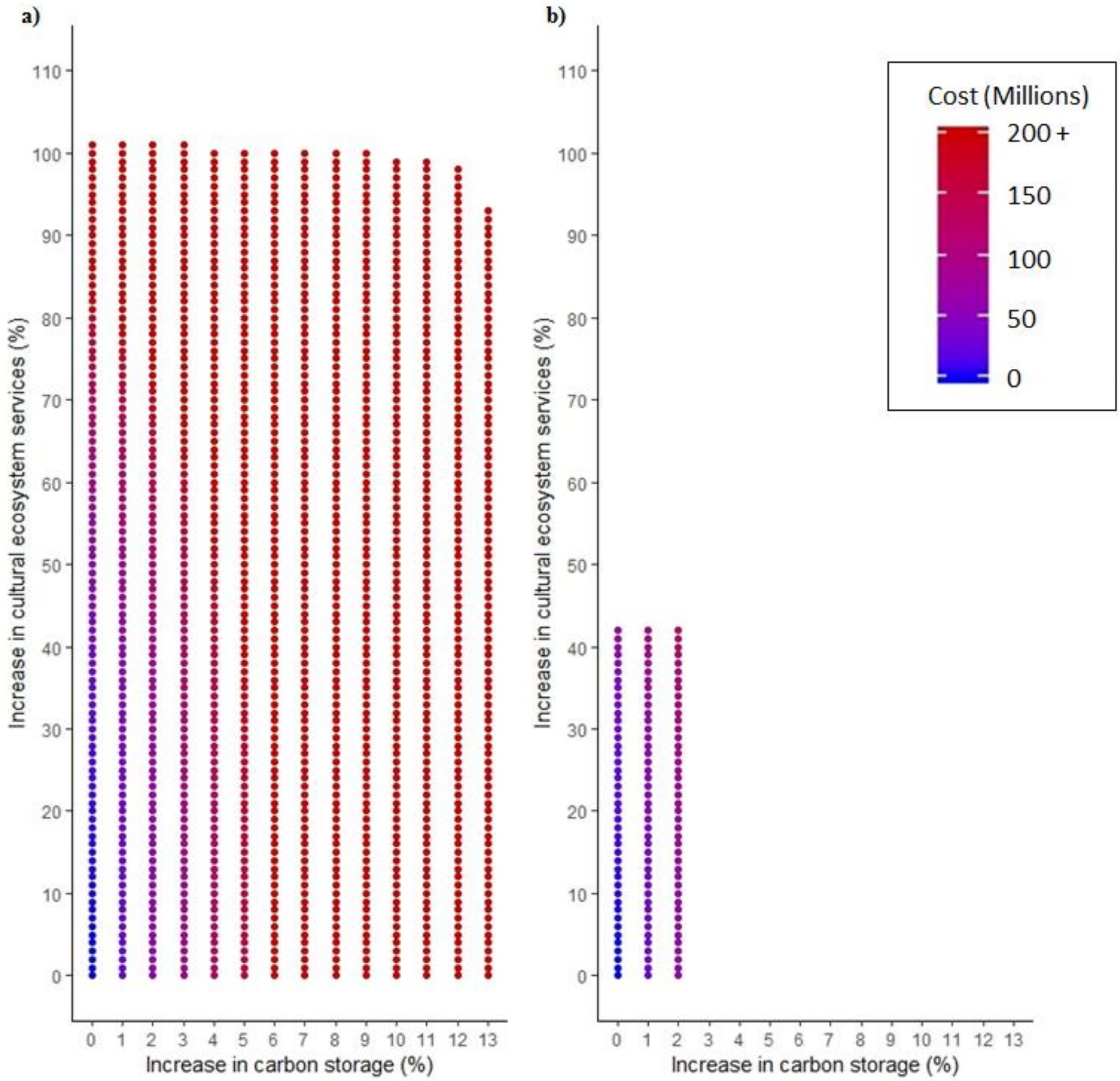

Figure 5.2 All feasible combined carbon storage and cultural ecosystem service targets that could be achieved and their associated management costs under a) the socially inequitable scenario, and b) the socially equitable scenario. The points represent each feasible target, and the percentage increase in carbon storage and the cultural services under this target. All costs are in AUD$.

The specific management actions implemented within each park differed between each scenario and across the ecosystem service targets. Under the socially inequitable scenario, when high targets were set for the cultural ecosystem services a large number of parks were allocated Management Action 7, where 80% of grass cover is replaced with tree cover and a combination of facilities is added to the parks (Figure 5.3). Furthermore, this Action was predominantly allocated to parks located in the outer regions of the Brisbane LGA. However, when high targets were set for carbon storage a large number of parks are allocated Management Action 1, where 80% of grass cover is replaced with tree cover, with no change in the facilities present. Under the socially equitable scenario, high targets for the cultural ecosystem services led to a large number of parks allocated Management Action 3 (add a playground) and 4 (add a footpath the width of the park) (Figure 5.4). High targets for carbon storage meant a larger number of parks were allocated Management Action 1 (replace 80% of grass cover with tree cover) and 6 (do nothing). There was also a more even distribution of the different management actions across the region under the socially equitable scenario (figure 5.4).

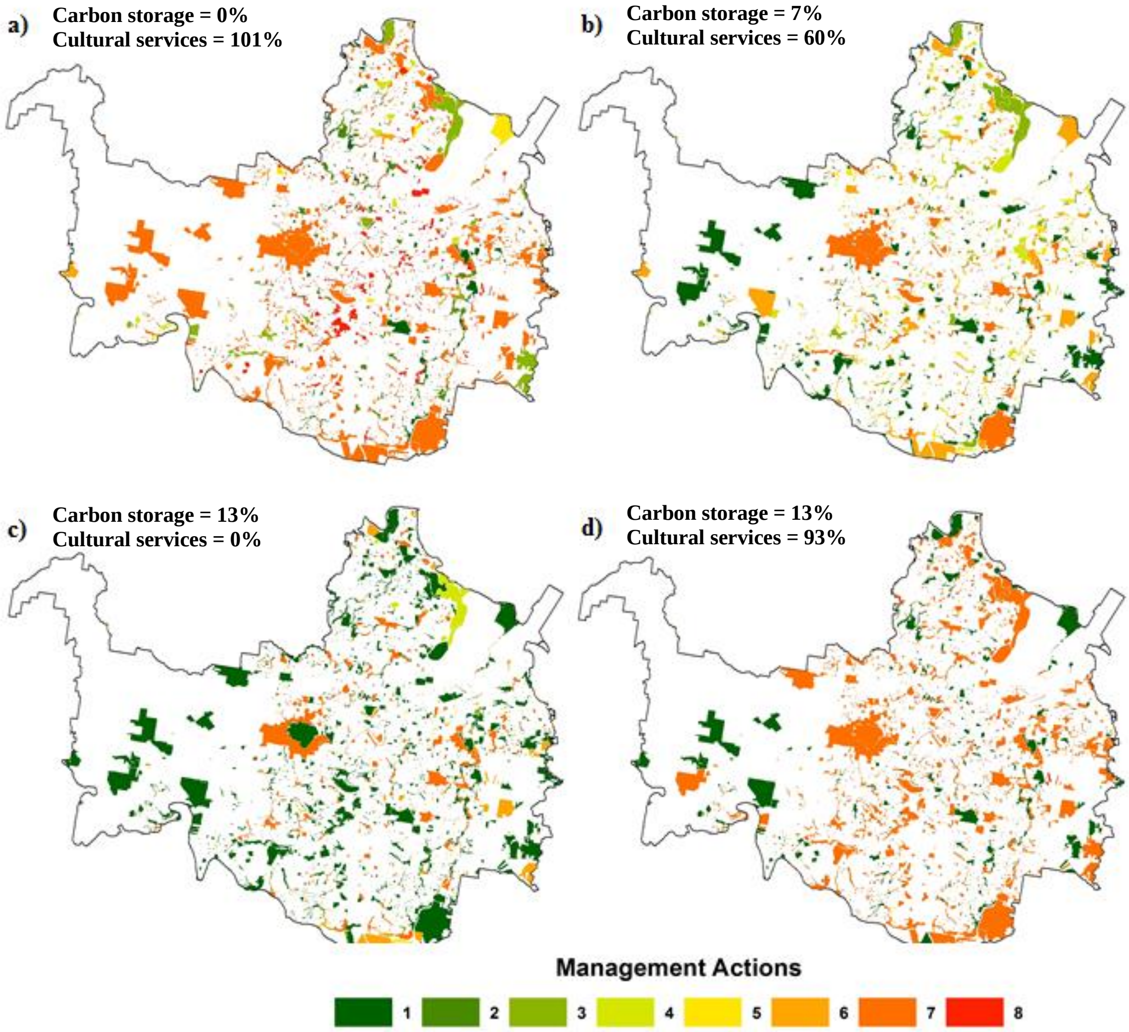

Figure 5.3 Spatial allocation of management actions for different ecosystem service targets under the socially inequitable scenario, with labels representing the ecosystem service targets. a) a 0% target for carbon storage with a 101% target for the cultural services, b) a 7% target for carbon storage with a 60% target for the cultural services, c) a 13% target for carbon storage and a 0% target for the cultural services, and d) a 13% target for carbon storage and 93% target in the cultural services.

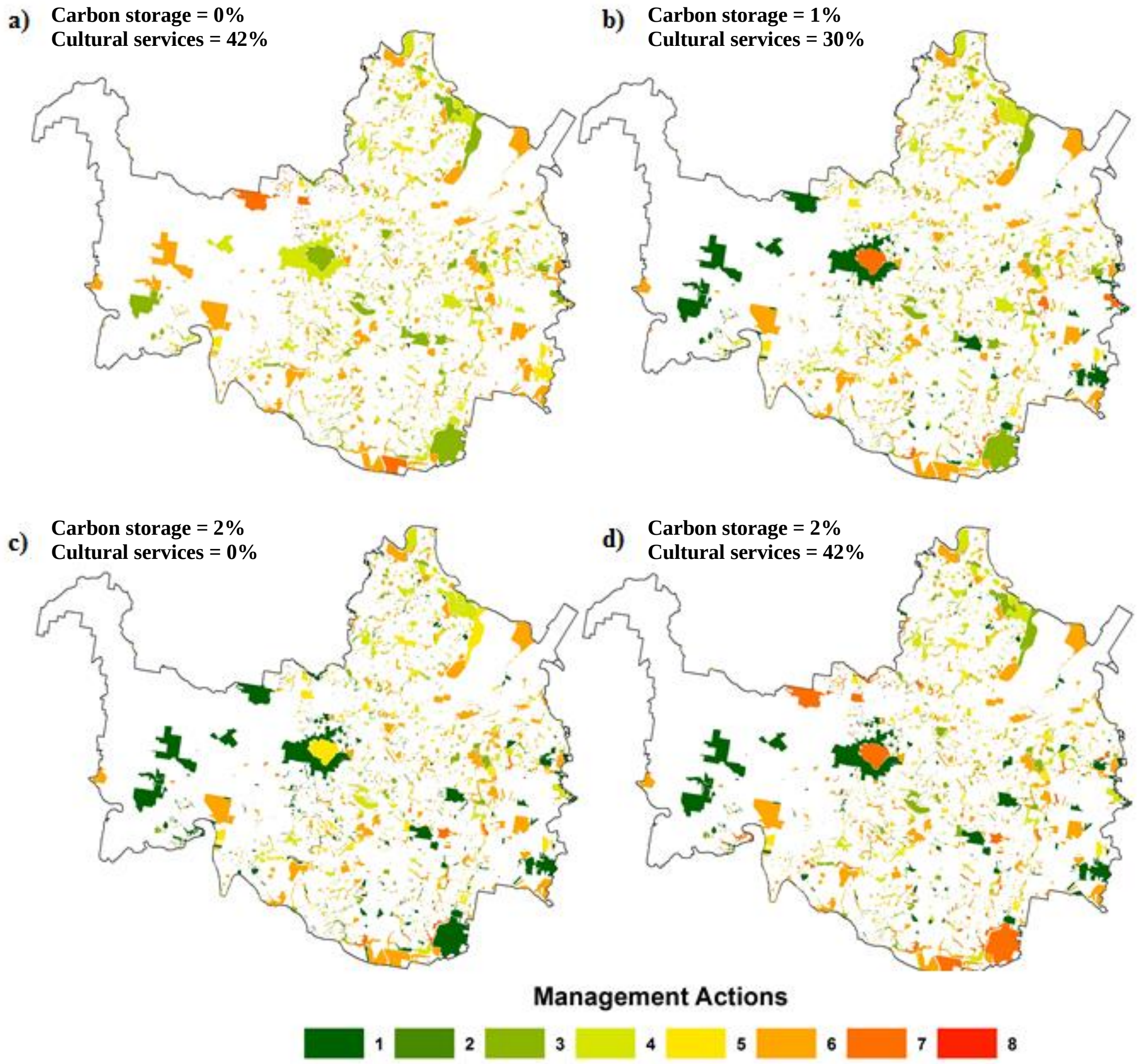

Figure 5.4 Spatial allocation of management actions for different ecosystem service targets under the socially equitable scenario, with labels representing the ecosystem service targets. a) a 0% target for carbon storage with a 42% target for the cultural services, b) a 1% target in carbon storage with a 30% target for the cultural services, c) a 2% target for carbon storage and a 0% target for the cultural services, and d) a 2% target for carbon storage and 42% target for the cultural services.

5.5 DISCUSSION

This study provides new insights into the optimal spatial allocation of multiple management actions across urban parks to achieve the provisioning of multiple ecosystem services. It also provides guidance on the consequences of aiming to achieve social equity; an important objective for urban environmental planning (Jennings et al. 2017; Rigolon 2016). My results emphasise that the implementation of multiple management actions are critical for the provision of multiple ecosystem services and this becomes even more so when equity is considered. However, accounting for social equity reduces the increases in ecosystem service provision that are feasible for each ecosystem service, and also increases management costs. This information is of particular importance for urban landscapes where there is a need for more effective management strategies to increase the provisioning of multiple services to accommodate the demand from growing urban populations (Andersson et al. 2015; Snäll et al. 2016).

There were striking differences in the optimal management actions to achieve targeted increases in carbon storage and the cultural ecosystem services. To achieve carbon storage targets, a larger number of parks were allocated Management Action 1. Management Actions 1 and 7 are the only actions that increase tree cover, and are therefore capable of increasing carbon storage (Davies et al. 2011). However, Management Action 1 was the cheaper of the two as it did not include adding facilities and was therefore most often allocated to minimise cost. However, to achieve targets for both carbon storage and the cultural ecosystem service, Management Action 7 was more commonly allocated as it increased tree cover and facilities, positively impacting both ecosystem services. These results highlight that park management strategies that consist of multiple actions are likely more capable of increasing multiple ecosystem services, but are also more costly. In a review of decision-making processes for ecosystem services, Martinez-Harms et al. (2015) found that only 19% of studies have systematically assessed how different resources should be allocated to different actions to manage ecosystem services. My study highlights the importance of achieving this for effective management of multiple ecosystem services in urban parks. I argue that the prioritization of management actions, and assessment of their costs and capabilities, should play a stronger role in the decision process when managing multiple ecosystem services.

Accounting for social equity in the spatial allocation of management actions reduced the targets that could be achieved for both services and increased management costs. The lower number of feasible targets for both ecosystem services in the socially equitable scenario were because of the spatial restrictions on where each ecosystem service improvements were required (Halpern et al. 2013). My study found that carbon storage, in particular, was highly restricted, with only a 2% target achievable under the socially equitable scenario, compared to the 13% target in the socially inequitable scenario. This was due to the fact that many SA3 regions included only a few parks and the management actions were unable to substantially increase carbon storage in these parks because of their size and high current tree cover. The effect of these spatial restrictions can also be seen in the spatial arrangement of the management actions. To achieve targets for carbon storage, a larger number of parks in the socially equitable scenario were allocated Management Action 6 (do nothing). This was due to these parks being within SA3 regions that had already reached the carbon storage and cultural ecosystem services target through management actions allocated to other parks in the SA3, and so these parks were allocated Action 6 to reduce costs. These results agree with studies focusing on the impact of equity on payments for ecosystem services and biodiversity conservation targets, which find that the most equitable options are usually not the most cost effective and restrict the targets that are achievable (Halpern et al. 2013; Narloch et al. 2011; Pascual et al. 2014). Furthermore, these results indicate that evenly distributing increases in carbon storage across parks to ensure equity in management burdens across the urban landscape will severely restrict the amount this ecosystem service can be increased by.

Ensuring social equity requires not only equal spatial distribution of the services, but also ensuring equal distribution of benefit from these services (Schröter et al. 2017). The social equity scenario used in this study focused on ensuring that the ecosystem service benefits provided by each park were evenly distributed across the park network. However, it did not take into account the benefits each individual person receives, and whether each person receives an equal percentage increase in benefits (Palomo et al. 2016). This was to simplify the decision problem and avoid spatial dependencies among actions in different SA3s (e.g., due to a person in one SA3 benefiting from an action taken in another SA3). As discussed in *Chapters 3* and *4*, people receive benefits from a wide range of parks that are not necessarily within the same SA3 region as they reside in. Therefore, a change in the provisioning of an ecosystem service in a single park could affect the benefits that people from all over Brisbane get from that park. This could mean that equally distributing increases in ecosystem service provisioning across each SA3 region could still result in inequitable distributions of increased benefits to urban residents. Furthermore, in this study I set spatial constraints on carbon storage in the socially equitable scenario. This was to ensure that increases in carbon storage were evenly distributed across the urban landscape, to prevent certain areas being burdened with maintaining this service. However, this is not always a concern across urban landscapes, where there is often a single governing body who maintains all urban public parks and therefore no one is burdened more than others (Ernstson et al. 2010; Lawrence et al. 2013). Therefore, it is often not necessary to place spatial restrictions on where increases in carbon storage can be achieved across the landscape. One way to approach this for future research is to first identify how the locations of the ecosystem services of interest can affect social equity within the landscape being studied. Using this information, the optimisation decision problem can be set up to place specific spatial targets for the services where social equity must be considered for, and global targets for the other ecosystem services. To determine which ecosystem services social equity is a concern for could require incorporating stakeholder engagement into the decision process when identifying ecosystem service targets (Arkema et al. 2015). This will ensure that appropriate targets will be set for ecosystem services that local communities and stakeholders believe should be equitably distributed across the landscape, and global targets set for all other services.

The models used to calculate the provisioning of the ecosystem services also assumed that the relationships between ecosystem service provisioning and the park characteristics remained the same regardless of how much the park characteristics change under each Management Action. While this is likely to be the case for carbon storage, which has a positive linear relationship with tree cover (Mitchell et al. 2018), it is possible the relationship between the cultural ecosystem services and the park characteristics is not constant. For example, increased grass cover may increase opportunities for recreation, and cultural services, within parks up to a certain amount, after which recreation decreases as grass cover continues to increase (Bjerke et al. 2006). This decrease could be due to there not being enough tree cover to provide patches of shade to conduct activities within, or a high proportion of grass cover is regarded as unattractive (Bjerke et al. 2006). Therefore, I recommend that further studies should focus on further untangling the relationships between cultural services and park characteristics to determine if these relationships change when park variables increase or decrease past certain thresholds. Incorporating this information into the spatial prioritisation of management actions could help provide management strategies that better reflect reality.

The use of spatial optimisation tools in the management of multiple ecosystem services will ensure more efficient and cost-effective management that ensure urban landscapes are multifunctional. Incorporating spatial optimisation tools into the decision-making process for managing multiple ecosystem services across landscapes can ensure more informed decision-making, and transparency about the implications and costs of different approaches or targets (Luck et al. 2012). Furthermore, it can help identify and prioritise resources and regions across landscapes to ensure the future provisioning of multiple ecosystem services. For example, within urban areas, it can help inform urban development and park management plans that aim to achieve targeted increases in the provisioning of multiple ecosystem services that should be distributed equitably across the urban landscape (Jennings et al. 2017; Rall et al., 2015). However, to achieve this, decision makers need to be aware of, and accommodate, the increase in management costs that achieving social equity can incur, and the potential restrictions in the ecosystem service targets that can be achieved.

5.6 CONCLUSION

We present a novel approach to spatially prioritising the allocation of multiple management actions to achieve targets in the provisioning of carbon storage and cultural services across an urban park network. My results show that it is not always possible to increase both ecosystem services simultaneously to a desired amount. Implementing management strategies that include multiple actions can help ensure that the provision of multiple ecosystem services increases simultaneously, but these actions are more costly. Furthermore, ensuring social equity in the distribution of the ecosystem services can further increase costs and reduce the feasible increases in ecosystem service provision. However, by better identifying the management actions available, and the optimal allocation of these actions, it is possible to increase the provisioning of multiple ecosystem services across urban landscapes while accounting for social equity.

CHAPTER 6
CONCLUSIONS

As human populations continue to grow and the demand for multiple ecosystem services increases, there is a growing need to manage multiple ecosystem services effectively across landscapes by implementing management actions that promote synergies rather than trade-offs between services (Bennett et al. 2015). This requires understanding the drivers and processes underpinning the provisioning of multiple ecosystem services (Raudsepp-Hearne et al. 2010; Turkelboom et al. 2018). Furthermore, it is important to understand when and where trade-offs and synergies occur among the ecosystem services, and how management actions and policies influence these relationships (Bennett et al. 2009). Ignoring these drivers and the relationships that result could lead to ineffective management strategies and perverse outcomes. However, to date, there remains a lack of knowledge on how to use these linkages between specific policies and drivers of ecosystem service trade-offs and synergies to inform the management of multiple ecosystem services.

To address this gap, my thesis aimed to evaluate the importance of understanding the drivers of ecosystem service change, and the resulting trade-offs and synergies, and apply this understanding to managing complex landscapes for multiple ecosystem services. To achieve this I used a literature review, a social survey, scenario analysis, and systematic conservation planning methods, using the urban landscape of Brisbane, Australia, as a case study. Specifically, I addressed four separate objectives: (i) Identify how often drivers and mechanisms linking drivers to ecosystem services are considered in assessments of synergies and trade-offs among ecosystem services and provide recommendations for improving these assessments (*Chapter 2*); (ii) Understand the drivers underpinning the provisioning of multiple cultural ecosystem services within an urban landscape (*Chapter 3*); (iii) Determine how trade-offs and synergies vary under different urban park management actions (*Chapter 4*); and (iv) Apply an understanding of the links between drivers and ecosystem services to maximise provisioning of multiple ecosystem services and achieve equity (*Chapter 5*).

In this concluding chapter, I synthesise the main findings from each of the chapters of my thesis, and discuss their implications for the management of multiple ecosystem services. I then identify the major contributions, discuss challenges and limitations, and recommend future research directions.

6.1 MAIN FINDINGS

6.1.1 Assessing ecosystem service trade-offs and synergies: the need for a more mechanistic approach. (*Chapter 2*)

The positive (synergistic) and negative (trade-off) relationships that exist among ecosystem services are influenced by drivers of change, such as management strategies and biophysical drivers (Bennett et al. 2009). Identifying the drivers underpinning ecosystem service relationships could be crucial in identifying the most effective strategies to manage multiple ecosystem services (Howe et al. 2014). However, the extent to which assessments of ecosystem service trade-offs and synergies use methods that explicitly consider and identify the drivers of these relationships is currently unknown. To determine how often drivers are explicitly considered, and the methods used to achieve this, in assessments of ecosystem services relationships (Objective 1), I conducted a systematic literature review. I found that the majority of assessments of ecosystem service trade-offs and synergies do not explicitly identify the drivers generating the relationships. There was also a strong link with the methods applied to identify the trade-offs and synergies, with assessments using more process-driven methods, such as scenario analysis and field experiments, more likely to identify the drivers and mechanisms underpinning the relationships. However, the majority of studies use methods incapable of this, such as correlation, and a failure to account explicitly for drivers can result in strong confounding factors. Furthermore, there was a stronger focus on some types of drivers of trade-offs and synergies being identified over others. Policy instruments were commonly identified drivers, but few cultural drivers, such as socio-political and religious factors, were considered in the articles reviewed. To ensure effective strategies are implemented to manage multiple ecosystem services, I recommend the stronger uptake of explicitly incorporating drivers into assessments of ecosystem service relationships, and a stronger socio-ecological approach to identify ecosystem services trade-offs and synergies and to inform management decisions.

6.1.2 Differentiating the effect of urban greenspace characteristics and socio-demographics on multiple cultural ecosystem services (*Chapter 3*)

As urban populations continue to grow, the demand for cultural ecosystem services within urban areas will also increase (Andersson et al. 2015; Rall et al. 2017). As discussed in *Chapter 2,* it is important to understand what drives the provision of these services to ensure the correct variables are targeted by management actions to effectively increase the provision of multiple urban cultural services (Kremer et al. 2016). However, we currently have limited understanding of what drives the simultaneous provision of multiple cultural services within urban greenspaces (Rall et al. 2017). To understand the drivers underpinning the provisioning of multiple cultural services within urban parks (Objective 2), I collected data on park visitation within Brisbane's parks for activities related to four cultural ecosystem services (opportunities for exercise, nature interactions, relaxation and social interactions), and assessed the influence of the characteristics of the park and the visitors on which parks were used for different cultural services. I found that use for all four ecosystem services were associated with spatial, environmental and park facility characteristics. Only opportunities for exercise and social interactions were associated with socio-demographic characteristics of the park visitors. In addition, the degree to which spatial, environmental and facility variables increased or decreased the rates at which parks were visited varied among the services. This indicates that park variables, including tree cover and presence of facilities, are influencing some cultural ecosystem services at a greater rate than others. I conclude that, by introducing management actions that target specific variables within urban parks, such as the type of facilities present or tree cover, it may be possible to increase the provision of multiple cultural ecosystem services simultaneously.

6.1.3 Urban ecosystem service trade-offs and synergies under different management scenarios (*Chapter 4*)

Effective management of urban greenspaces to deliver multiple ecosystem services requires an understanding of the trade-offs and synergies between the services under different management strategies (Gaston et al. 2013). There are a variety of methods available to identify trade-offs and synergies between ecosystem services (Mouchet et al. 2014), but, as demonstrated in *Chapter 2*, they vary in their ability to identify how relationships vary under different drivers. In order to determine how trade-offs and synergies vary under different urban park management actions (Objective 3), I first used a spatial correlation approach to identify spatial trade-offs and synergies between multiple urban ecosystem services (opportunities for exercise, nature interactions, relaxation, social interactions and carbon storage) within Brisbane's parks. I then used a scenario analysis to determine how those relationships vary under different urban revegetation management strategies. This approach demonstrated that trade-offs and synergies between the ecosystem services do vary under the different management strategies. The spatial correlation analysis identified trade-offs between carbon storage versus opportunities for relaxation and social interactions, a synergy between carbon storage and nature interactions, and synergies between all the cultural services. However, the scenario analysis revealed that under revegetation strategies that involved increasing tree cover a synergy remained between carbon storage and nature interactions, but trade-offs occurred between nature interactions versus the remaining cultural services. No significant trade-offs or synergies were identified under the management strategy that focused solely on increasing mid storey vegetation. This variation in ecosystem service relationships, identified using the scenario analysis, suggest the limitations of using a standard spatial correlation that does not control for drivers of ecosystem services. These results support my findings in *Chapter 2* that accounting for drivers are important for the effective management of multiple ecosystem services and to determine which actions promote synergies rather than trade-offs between services.

6.1.4 Management priorities for equitable distribution of urban ecosystem services (*Chapter 5*)

When identifying which management actions to implement within urban landscapes to increase the provisioning of multiple ecosystem services it is important to choose a variety of different actions, as they can increase different ecosystem services (as demonstrated in *Chapter 4*). Ideally, different actions should be spatially allocated across landscapes to, not only ensure maximum increases in multiple services, but also to ensure social equity in access and management (Hansen and Pauleit 2014). In this chapter I applied my models characterising the links between drivers and ecosystem services to prioritise multiple actions for multiple services to maximise the provisioning of multiple ecosystem services and equity (Objective 4). To do this, I spatially prioritised the implementation of different management actions to increase the provisioning of multiple services to specific targets across Brisbane’s urban park network and compared these results and their associated costs when social equity was considered and was not considered. I found that accounting for social equity reduced the aggregate amount each ecosystem service could be increased by, and increased the costs of achieving these increases. Furthermore, I found that management actions that involved no actions were more commonly applied when social equity was considered. This is due to the restrictions social equity places on where ecosystem services are required to be provided. When social equity was not considered, management actions that consisted of multiple actions were more commonly implemented as they increased both ecosystem services. These results provide information on how to spatially prioritise multiple actions for multiple services. When combined with identifying drivers (*Chapter 3*) and the trade-offs and synergies under different management actions (*Chapter 4*), the information in this chapter can help inform the efficient design of park management to achieve target increases in multiple ecosystem services. This approach could be further applied across a wide range of landscapes to ensure optimal provisioning of multiple ecosystem services.

6.2 MAJOR CONTRIBUTIONS

My thesis is multidisciplinary, drawing on methods and theory from landscape ecology, human geography, social science, decision science, and environmental management to advance our knowledge on managing multiple ecosystem services. Specifically, I focus on understanding how to identify the most effective management strategies while accounting for trade-offs and synergies between ecosystem services. My findings are relevant to ecologists, practitioners and policy-makers with interests in designing and managing sustainable landscapes that benefit both human wellbeing and ecosystem health. I performed a systematic literature review and conducted original research using empirical data. By doing this, I was able to identify the main knowledge gaps, test hypotheses and evaluate the role of drivers in the effective management of multiple ecosystem services. The major contributions of my thesis are described in the following subsections.

6.2.1 The fundamentals of ecosystem service relationships

Previous studies have conceptualised the importance of understanding the drivers underpinning ecosystem service relationships to inform better management of multiple services (Bennett et al. 2009; de Groot et al. 2010; Howe et al. 2014). In *Chapters 1* and *2* I analysed the literature to conceptually outline how drivers can generate trade-offs and synergies, and to demonstrate that by treating management actions as drivers of ecosystem service relationships we can potentially implement more effective management of multiple ecosystem services. The findings of my original research chapters support this, as I found that understanding the drivers underpinning the provisioning of multiple cultural ecosystem services can help predict when and where trade-offs and synergies may occur between services, potentially allowing for better management of multiple services (*Chapter 3*). Furthermore, I found that explicitly incorporating drivers into assessments of ecosystem services trade-offs and synergies can increase our understanding of how management actions will affect the provisioning of multiple services (*Chapters 4* and *5*). These findings provide evidence that trade-offs and synergies between ecosystem services can change depending on the drivers, such as management actions, present. Therefore, decision-makers and policy-makers need to think of management actions as drivers of trade-offs and synergies, rather than just as solutions to increase ecosystem services, and explicitly consider the drivers of ecosystem service relationships when implementing strategies to manage multiple ecosystem services.

6.2.2 Identifying management actions as drivers of trade-offs and synergies

Although previous studies have used scenario analyses to identify ecosystem service trade-offs and synergies (e.g., Briner et al. (2013), Lauf et al. (2014), and Fezzi et al. (2015)) these have not used empirical data to conduct a time for space swap analysis to determine if ecosystem service trade-offs and synergies occurring spatially change over time under different management actions (*Chapter 4*). By identifying the trade-offs and synergies occurring between a group of ecosystem services using a spatial correlation approach and a scenario analysis, I showed that management actions implemented based on the findings of methods that only identify the trade-offs and synergies occurring spatially (such as spatial correlation) are likely to lead to perverse outcomes, and potentially unexpected declines in ecosystem services. Scenario-based approaches to identify ecosystem service relationships provide crucial information on the trade-offs and synergies generated by different management strategies over time. These findings should help managers assess the effectiveness of different actions for managing multiple ecosystem services. I therefore recommend that methods able to capture the temporal changes in ecosystem services be used to determine the actions to implement to assess trade-offs. However, as outlined in *Chapter 2*, this will require the careful consideration of the type of data required to conduct these methods. For example, a scenario analysis requires data on the ecosystem variables underpinning the provisioning of the ecosystem services (*Chapter 4*), whereas a spatial correlation approach only requires measuring the quantity of each ecosystem service at given locations (Mouchet et al. 2014). However, this data will ensure management actions are implemented that avoid trade-offs occurring and ensure the effective management of multiple ecosystem services.

6.2.3 The spatial prioritisation of ecosystem service management actions

Previous studies have used spatial optimisation tools to prioritise areas for ecosystem service management (Chan et al. 2006; Schröter and Remme 2016), and assess the impacts of different land use strategies on ecosystem service provisioning (Law et al. 2017). My thesis contributes to this knowledge by optimising the spatial allocation of a variety of management actions across a landscape to achieve increases in multiple ecosystem services. With different management actions having the ability to improve outcomes for different ecosystem services, and by different amounts (*Chapters 3* and *4*), spatial optimisation tools are capable of identifying the optimal allocation of different management actions to achieve target increases in the provisioning of multiple services (*Chapter 5*). These findings from my thesis also highlight the applicability of using optimisation tools for managing ecosystem services and ensuring landscape sustainability, where environmental and economic benefits are achieved alongside social equity (Wu 2013). Furthermore, these results highlight that, instead of implementing a single management action to increase ecosystem service provisioning, it is in fact more effective to identify a variety of management actions and optimise their allocation across the landscape. These findings suggest that this use of optimisation decision tools can be applied to managing multiple ecosystem services not just across urban landscapes, but a wide range of natural and human modified landscapes to increase the sustainability of these landscapes.

6.2.4 Understanding urban cultural ecosystem service drivers and relationships

There remains limited information on the factors affecting where people go to receive cultural ecosystem services within urban landscapes, and why (Gómez-Baggethun and Barton 2013; Rall et al. 2017). I demonstrate that socio-demographic characteristics of people can drive the provisioning of social interactions and exercise within urban landscapes, but that the environmental and facility characteristics of urban parks influence a wider variety of cultural ecosystem services (*Chapter 3*). Much of the research focusing on cultural ecosystem services in urban landscapes focuses on the social factors affecting service delivery (Shan 2014; Wilkerson et al. 2018; Zhang et al. 2013). However, my findings indicate social factors are not relevant to all cultural urban ecosystem services. Therefore, to manage multiple cultural ecosystem services within urban landscapes it is recommended that policy-makers focus on altering park characteristics to increase the amount of each service they provide, rather than socio-demographic characteristics. Furthermore, I showed that both trade-offs and synergies occur between cultural services, and are influenced by the characteristics of urban greenspaces. Opportunities for nature interaction tends to trade-off with opportunities for exercise, relaxation and social interactions due to the preference for different types of vegetation structure (*Chapters 3* and *4*). However, synergies predominantly exist between opportunities for exercise, relaxation and social interactions within urban greenspaces. This information provides crucial information for planning urban greenspaces to provide multiple cultural ecosystem services to accommodate growing urban populations (United Nations 2015).

6.2.5 The impacts of social equity on ecosystem service management

I demonstrated the implications of considering social equity when spatially optimising the allocation of management actions to increase the provisioning of multiple urban ecosystem services (*Chapter 5*). This is a significant advance for the management of multiple ecosystem services, as previous studies have only looked at social equity for a single ecosystem service (Jenerette et al. 2011; Martinez-Harms et al. 2018), but no previous study has incorporated social equity into spatial optimisation of management strategies to increase multiple ecosystem services simultaneously. My findings demonstrate that policy-makers are likely to face a trade-off when it comes to achieving increases in multiple ecosystem services at minimum costs, and achieving social equity in the management of these services across the landscape. To better manage this potential trade-off, it is recommended that the decision-making process to manage multiple services should include the careful consideration of the pros and cons of accounting for social equity given the costs. It is also important that these discussions on social equity are conducted as a transparent process and includes stakeholder engagement, to prevent discord and unequal representation between different social groups in the final decision (Dawson et al. 2017).

6.3 LIMITATIONS AND FUTURE RESEARCH

This thesis conceptualises and demonstrates the management of multiple ecosystem services in complex landscapes. In this section, I discuss the main limitations of this thesis, and suggest future research directions to advance our knowledge on this topic.

6.3.1 Correlation does not imply causation

To explicitly identify the drivers underpinning ecosystem service relationships, and the mechanisms linking the drivers to the delivery of ecosystem services, ideally requires methods capable of identifying causal mechanisms. This means developing hypotheses about the causal links between variables while controlling for confounding factors in the sampling and statistical design (Law et al. 2017; Pearl 2009; Winship and Morgan 1999). In *Chapters 3, 4* and *5* I used a regression approach to identify trade-offs and synergies between ecosystem services, in the form of Poisson zero-inflated models developed to calculate ecosystem service provisioning. This approach can identify the associations between variables that underpin these relationships, predict the impacts of management actions on these relationships and, in general, is a step forward from standard spatial correlation or overlap analysis (Mouchet et al. 2014). However, I still rely simply on associations between variables rather than explicitly identifying causation (Mouchet et al. 2014). Without the use of a causal approach there remains the risk that confounding variables are present that are influencing the results of the analysis. This reduces the ability for the models developed in this thesis to reflect reality (Law et al. 2017).

Causal based approaches that could be applied to identify trade-offs and synergies include field experiments, causal inference, simulation models and process-based statistical models (Law et al. 2017; Mouchet et al. 2014). Using a causal approach, such as causal inference, to identify relationships between ecosystem services in multifunctional landscapes is challenging due to a number of factors. The number of variables present that affect ecosystem services, and the mechanisms linking them to ecosystem service delivery, can be very large and their interactions can be highly complex (Sugihara et al. 2012). Therefore, developing clear hypotheses and controlling for these variables within the sampling and statistical design is difficult to achieve (Sugihara et al. 2012). Furthermore, it can require a significant amount of data, which is often difficult to obtain due to logistical or resource constraints, and careful consideration of the data to avoid bias in which variables are having a larger influence on the ecosystem services (Law et al. 2017). To allow for more causal approaches, future data collection strategies should focus on first identifying the variables potentially affecting ecosystem service provisioning and then framing data collection strategies around the variables of concern. This includes ensuring data is collected at the appropriate scale, and the correct type of data is collected that can be incorporated into a causal approach (Bagstad et al. 2018). Furthermore, as highlighted in this thesis, it is important that data collection strategies are chosen that are capable of collecting both social and ecological data, which are both necessary to understanding the drivers underpinning ecosystem service delivery. Collecting the appropriate data to allow for more causal-based approaches to assessing ecosystem service relationships is an important direction for future research.

6.3.2 Social and cultural drivers of ecosystem service relationships

There is a large quantity of literature discussing the social factors that influence ecosystem service provisioning (Daw et al. 2015; Martín-López et al. 2012; Riechers et al. 2018). However, as identified in *Chapter 2*, human (or social) drivers of ecosystem service relationships are often not considered in the literature. I began to uncover some of the social variables underpinning cultural ecosystem service trade-offs and synergies in *Chapter 3*, but there are likely to be other social variables that influence the relationships between ecosystem services. Future research should focus on incorporating the social factors identified in the literature as driving ecosystem service provisioning to inform assessments of trade-offs and synergies.

Cultural and religious values can potentially play a large role in the relationships occurring between ecosystem services (Daw et al. 2015). Furthermore, socio-political drivers, such as gender equality and even systems of government can also influence trade-offs and synergies between ecosystem services (Berbés-Blázquez et al. 2016). To ensure effective ecosystem service management, further research is required to understand how these social and cultural factors can influence ecosystem service relationships. This includes understanding how social factors affect the trade-offs and synergies occurring between different groups of ecosystem services, and which social factors should policy-makers focus on to effectively manage multiple services. Approaches to achieve this could include comparable assessments between different social groups or socio-political environments, as well as empirical assessments of ecosystem service relationships considering a wider number of social variables. This information would provide vital information on how to effectively manage multiple ecosystem services across culturally diverse landscapes.

6.3.3 Incorporating flow and demand into ecosystem service management

Effective management of multiple ecosystem services across landscapes should ensure that those who require these ecosystem services have access to them (Wolff et al. 2015). Therefore, management should consider the demand for the services, and movement or ecosystem service flow between where the demand is located, and where ecosystem services are supplied (Mitchell et al. 2015; Wolff et al. 2017). In *Chapter 5*, I illustrated the impact that ensuring social equity in management and access to ecosystem services can have on the management of multiple ecosystem services within urban landscapes. However, this analysis only focused on equally distributing increases in ecosystem services across the landscape, and did not focus on where the demand for these service were located, and the movement (or flow) from this demand to where the ecosystem services were being provided. Furthermore, regions often consist of different dominant socio-demographic groups that can influence which ecosystem services are in demand in different areas (Wilkerson et al. 2018). Therefore, the distribution of ecosystem services determined using this approach may not accurately reflect where the demand for the ecosystem services are, and this could lead to the implementation of management actions where some areas may receive ecosystem services that are not in demand, while other areas may not receive enough. This could lead to unequal distribution in ecosystem service benefits across the landscape.

Alternative methods and further development of existing approaches are required to better incorporate demand and flow into the management of multiple ecosystem services and ensure social equity in the provisioning of ecosystem services (Schröter et al. 2017). Approaches that focus on equally distributing the benefits rather than the supply of ecosystem services across a landscape may help ensure better social equity. This could be achieved by determining where each person across the landscape receives their ecosystem service benefits from, and ensuring increases in ecosystem services reflect this (Martin et al. 2018). For example, if most people visit a particular park to receive an ecosystem service, the provisioning of this service should be increased here more than other parks, as more people benefit from this park (Jennings et al. 2017). Incorporating this information into the management of ecosystem services could potentially help ensure greater social equity in both the access and management of ecosystem services. Further research is required into uncovering the linkages between socio-demographic characteristics and ecosystem service demand, and incorporating ecosystem service flow into the spatial prioritisation of management strategies for multiple ecosystem services to ensure social equity.

6.4 CONCLUDING REMARKS

Managing ecosystem service trade-offs and synergies is critical to ensure and improve the provision of multiple services and the mental and physical benefits they provide to people. Developing new approaches to assess these relationships and understand the drivers that underpin them is vital to ensure we are capable of identifying the most efficient management strategies to manage multiple services. My thesis advances our understanding of how to manage multiple ecosystem services across landscapes, provides recommendations to more efficiently manage ecosystem services, and demonstrates ways forward for future research.

CHAPTER 7

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APPENDIX A: AN EVALUATION OF PARTICIPATORY MAPPING METHODS TO ASSESS URBAN PARK BENEFITS

This section is reproduced from the following paper, with some alternatives to formatting and structure:

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The idea for the manuscript was conceptualised by Greg Brown. Jonathan Rhodes and myself. The data was collected by Greg Brown, Jonathan Rhodes and myself. The analysis of the data was conducted by Greg Brown. The manuscript was written by Greg Brown, with editorial input from Jonathan Rhodes and myself.

A.1 ABSTRACT

Traditional urban park research has used self-reported surveys and activity logs to examine relationships between health benefits, park use, and park features. An alternative approach uses participating mapping methods. This study sought to validate and expand on previous participatory mapping research methods and findings and address spatial scaling by applying these methods to a large urban park system. Key challenges for spatial scaling included ambiguity in park classification and achieving representative sampling for larger and spatially-disbursed urban residents. We designed an internet-based public participation GIS (PPGIS) survey and used household and volunteer sampling to identify the type and locations of urban park benefits. Study participants (n=816) identified locations of physical activities and other urban park benefits (psychological, social, and environmental) which were analyzed by park type. Consistent with previous suburb-scale research, we found significant associations between urban park type and different urban park benefits. Linear parks were significantly associated with higher intensity physical activities; natural parks were associated with environmental benefits; and community parks were associated with benefits from social interaction. Neighbourhood parks emerged as significantly associated with psychological benefits. The diversity of park activities and benefits were positively correlated with park size. Distance analysis confirmed that physical benefits of parks were closest to participant domicile, while social and environmental benefits were more distant. These results validate previous suburb-scale findings despite greater variability in park types and sampled populations. Future urban park research using participatory mapping would benefit from greater effort to obtain participation from under-represented populations that can induce nonresponse bias, and analyses to determine whether system-wide results can be disaggregated by suburb or neighbourhood to address social inequities in urban park benefits.

A.2 INTRODUCTION

Urbanization is a dominant global trend with over half the world's population now living in cities (United Nations, 2015). Urban parks and greenspaces are widely held to contribute to human well-being and quality of life (Chiesura 2004; Larsen et al. 2016), but the empirical evidence for the link between human well-being and urban green space is weak due to poor study design, confounding effects, bias or reverse causality, and weak statistical associations (Lee and Maheswaran 2011). The diversity and variability in urban populations, in combination with the heterogeneity of urban physical environments, make assessing urban greenspace benefits challenging. Urban design and planning outcomes that provide for parks and conserve greenspaces appear broadly justified based on *perceived* benefits, but parks and greenspaces do not contribute equally to the collective benefit enjoyed by urban inhabitants. In many cases, physical, psychological, and social health benefits appear inequitably distributed across urban populations (Jennings et al. 2016). Further, perceived access to urban parks (Wang et al. 2015a) or a favorable orientation to nature (Lin et al. 2014) appear more important than geographic access or proximity in predicting urban park use.

A variety of social research methods have been used to examine the putative benefits of urban parks and greenspaces. Participatory mapping methods, alternatively called public participation GIS (PPGIS), participatory GIS (PGIS), or volunteered geographic information systems (VGI), are increasingly used as a social research tool to assess the multiple benefits of urban parks and greenspaces. These methods offer an alternative to self-reporting surveys, activity logs, and direct observation methods such as SOPARC (McKenzie 2005) for identifying the public health benefits from park activities (Brown et al., 2014). Further, these participatory mapping methods have the flexibility to identify broader social values and cultural ecosystem services associated with urban greenspaces (Tyrväinen et al. 2007; Ives et al. 2017; Rall et al. 2017; Ribeiro and Ribeiro 2016).

Participatory mapping methods for assessing urban park and greenspace benefits have multiple threats to research validity. Some of the key validity issues for the spatial mapping of benefits include the variables/constructs being mapped, spatial scale of the study area (e.g., park, suburb, or entire urban area), physical landscape variability (e.g., water, vegetation, topography), park/greenspace facilities/amenities, distance from domicile, accessibility, park/greenspace classification, and population sampling representativeness. To date, these methodological issues have not been comprehensively addressed within the same study, with reported studies examining a subset of these research issues.

In this study, the research objectives are to: (1) assess whether findings about the distribution of park benefits (physical, environmental, psychological, social) identified in previous participatory mapping studies that were limited in scope and scale are applicable to a large, diverse urban park system; and (2) examine the methodological challenges for scaling-up participatory mapping methods to assess urban park benefits in a large urban park system.

A.2.1 Review of related participatory mapping research

Brown et al. (2014) examined the distribution of urban park benefits (physical, psychological, social, and environmental) by park type using a park classification system developed by the

National Recreation and Parks Association (NRPA) (Mertes and Hall 1996). The study relied on a predominantly volunteer sample of urban residents (n=242 participants) living in one suburb in the larger urban area of Adelaide, Australia. The study found that different urban park types provide opportunities for physical activities with differential health benefits. Linear parks provided the greatest overall physical benefit while other park types provided important psychological, social, and environmental benefits. Distance to park was not a significant predictor of physical activity but park size was related to benefits with larger parks providing greater and more diverse benefits. The potentially confounding variables of park accessibility, park amenities, and physical landscape characteristics were not examined.

Ives et al. (2017) implemented a PPGIS study in four urbanising suburbs in the Lower Hunter region of NSW, Australia, and requested residents (n=418 participants) to identify important values of greenspace. The analyses examined the relationship between mapped values to physical landscape characteristics and also evaluated a simple greenspace classification typology (general, natural, sportsfield). The most frequently mapped value was physical activity and the majority of mapped values reflected positive attributes of greenspaces. Significant predictors for multiple greenspace values were distance to water and suburb identity, while the greenspace category was not significantly related to mapped values.

Rall et al. (2017) examined patterns of perceived cultural ecosystem services (CES) in the city of Berlin mapped by residents using convenience sampling (n=562 participants). The study examined the distribution of CES by land cover classification. About three-quarters of all CES were mapped in urban greenspaces or forests. The study found spatial differentiation of perceived cultural ecosystem services (CES) in greenspaces where the density of CES decreased from the inner to the outer edges of the city. Recreation, social, cultural heritage, and identity services were concentrated more heavily in the inner-city, while biodiversity, spiritual, inspirational, nature experience and educational services were more spatially scattered.

Bijker and Sijtsma (2017) examined whether greenspaces at different distances are important for the wellbeing of urban dwellers. The study focused on urban residents drawn from internet panels in three countries (Germany, Denmark, Netherlands: n=3,763 respondents). Participants were asked to identify natural places that were attractive, valuable, or important at four different spatial scales: local, regional, national, or world. The attractiveness of natural places increased with spatial scale while local natural places were visited most frequently. As the spatial scale expanded from the local area, more greenspace qualities were identified. At all spatial scales, “green nature”, recreation, and water qualities were the most frequently identified. Urban residents appear to have a “portfolio” of favorite places at multiple scales with local places being less special, but visited more frequently to counterbalance the stressful effects of population density. Places at the local and regional level especially provided opportunities for physical and social activities.

Pietrzyk-Kaszyńska et al. (2017) used participatory mapping to assess the non-monetary values of greenspaces in three cities in Poland. The study relied on sampling of volunteer participants (n=1640) who identified important urban greenspaces on a map, both formal and informal greenspaces, and who provided qualitative statements for their importance. The study found between 17% and 41% of places where respondents spent time were areas outside of formal greenspaces that were valued for their greenness, pleasant views, uniqueness, wild character and natural habitats. The findings highlighted the need to identify and include informal greenspaces in urban spatial planning and governance.

With the exception of the Brown et al. (2014), these studies assessed park benefits indirectly through measurement of landscape values, ecosystem services, or park qualities, and none of the studies implemented both household and voluntary/convenience samples in the recruitment of study participants. The novelty of this research is the direct measurement of urban park benefits in a large urban park system using participatory mapping methods, the inclusion of multiple sampling methods to evaluate potential bias and representativeness, and the identification of park classification issues when applying the methods to a large urban park system.

A.2.2. Study purpose and research questions

This study seeks to advance knowledge about the strengths and limitations of participatory mapping as a social research method for identifying urban park benefits in a large urban park system. We follow the initial design of Brown et al. (2014) who identified urban park activities and benefits (physical, psychological, social, and environmental) by park type in a study of a suburb in Adelaide. However, this study is more than a replication study and contains new research design innovations in addition to addressing the important issue of methodological scaling by applying the participatory mapping process to a large urban area and park system located in Brisbane, Australia (est. pop. 1.2 million). The key challenges for scaling-up from suburb to large urban park system include the ambiguity in park classification resulting from a greater diversity in parks and reserves across the system and sampling for larger and more heterogeneous human populations.

The first study innovation was to simplify the list of park activities to assess physical health benefits based on metabolic equivalent of task (MET). Metabolic equivalents are a unit used to estimate the metabolic cost of physical activity, with the value of one MET being approximately equal to an individual's resting energy expenditure (Jette et al. 1990). METs can be estimated for a range of physical activities based on the nature and the intensity of engagement in the activity. Park activities that could be mapped ranged from low energy, sedentary activities such as sitting, to higher energy activities such as running, cycling, and playing sport. The list of activity markers included new activities not previously used (*dog walking*, *water-based activities*, and *supervising children in parks*). As a design trade-off for simplicity in mapping, multiple MET levels (e.g., high, medium, low) were not provided for each activity as in the previous study even though most activities have varying MET intensity levels.

A second innovation was an effort to capture the frequency and duration of the mapped park activity to capture information about MET levels. The intent was better estimate the physical benefits associated with the activities. A third innovation was adapt and modify the NRPA (Mertes and Hall 1996) park classification typology criteria to the operational demands of larger, variable, and more complex urban park system.

Thus, this study seeks to answer research questions about the applicability of suburb-level findings about park benefits to a large urban park system as well as methodological questions about scaling-up the participatory methods.

The following research questions assess the distribution of park benefits within a large urban park system:

(1) What types of parks/reserves offer more (less) physical health benefits in an urban park system?
(2) Can the mapping of physical activities based on assumed MET levels provide reliable estimates of physical health benefits from different types of parks?
(3) How are multiple park benefits (environmental, physical, psychological, and social) distributed by park type and which types of parks offer disproportionately more (less) of these benefits?
(4) Does the diversity of park activities and benefits differ by park type and size?

(5) Is the distribution of physical activities and benefits related to distance from domicile?

The following research questions identify key issues in scaling-up participatory mapping methods to a large urban park system:

(6) How do population sampling methods (household vs. voluntary) in participatory mapping influence demographic and geographic representativeness of findings about urban park benefits?
(7) What geographic and social factors should be considered in classifying and analysing urban parks by park type for examining the distribution of benefits in a large and diverse system?

Following the answers to these questions, we discuss the strengths and limitations of participatory mapping as social research method for identifying urban park and greenspace benefits and how the method can be better applied to inform urban greenspace management.

A.3 METHODS

A.3.1. Study location

The geographic setting for this study was Brisbane, Australia, the capital city of Queensland with an estimated greater metropolitan area population of 2.35 million people. The Brisbane local government area (LGA), the physical boundary for this study, has an estimated population of 1.2 million and encompasses 1,338 km^2 (ABS 2015). The Brisbane City Council (BCC) manages the hundreds of parks and reserves located in the LGA that range in size from small neighborhood parks to large district parks, including two botanic gardens.

A.3.2. Sampling and data collection

The data collection portion of study was completed between October 2016 and January 2017. Two sampling methods were used to recruit participants to the internet-based participatory mapping (PPGIS) study:

Random household participants. Residential mailing addresses for the Brisbane City Council LGA were obtained from a commercial vendor (yell123.com). A total of 5,000 household addresses were randomly sampled from the address database stratified across suburbs with weightings proportional to the area of each suburb. A letter of invitation to participate in the study was sent on October 7, 2016 with a follow-up reminder postcard sent on October 18, 2016. An additional 2,500 household addresses were randomly selected using the same protocol as above and sent recruitment letters on October 24, 2016. No additional follow-up reminders were sent to this latter sample. Responses from this household sampling group were tracked by unique access code. To encourage participation, an incentive was offered consisting of a $10 gift voucher to a grocery/department store chain located throughout the greater Brisbane area. Alternatively, participants could select from one of three local charities who would receive a $10 donation on the participant's behalf at the close of the study.

Volunteer participants. The BCC sent an announcement of the study to community groups with potential interests in BCC parks via the *Greenheart Newsletter* mailing list. Community groups also advertised the survey through their own social networks, via Twitter and Facebook. The announcement contained the URL address of the study website. Volunteer participants were assigned different access codes from the household sample and tracked separately and were not offered an incentive for participation.

A.3.3. PPGIS methods and process

The research team developed an initial PPGIS survey based on previous research by Brown et al. (2014) and met with BCC professional staff responsible for park/reserve management to refine the list of activities and benefits to be included in the study. The survey was pre-tested with a convenience sample of colleagues of the research team and with BCC staff.

The PPGIS survey website contained four primary components: (1) an initial screen for study participants to enter their supplied access code (household sample) or to request a dynamic access code (volunteer sample); (2) a screen to obtain informed consent; (3) customized Google® maps interface instructing the participant to drag and drop different digital markers onto a map of the Brisbane LGA area; and (4) a set of text-based survey questions that followed the mapping activity. The digital markers for mapping activities and benefits were located in panels on the left of the screen where participants would drag and drop markers onto the map location representing the activity or benefit. The first panel consisted of 12 physical activities commonly associated with parks and greenspaces and the second panel consisted of 12 potential park benefits.

The physical activities were identified and selected to provide a range of physical activities for assignment to a metabolic equivalent of task (MET) category based on an assumed level of energy expenditure for the activity. Because a given activity (e.g., walking) can be done at multiple intensity levels, we made an assumption about the most common level of intensity associated with the activity for classification into the nominal categories of *high*, *medium*, or *low* energy expenditure. For example, walking activity can be done at multiple intensity levels (walking speeds) with estimated MET levels ranging from about 2 to over 5 (Jette et al. 1990). In this study, walking activity was classified as a *moderate* level MET activity while resting/sitting was classified as a *low* MET activity. The 12 physical activities and their assigned MET nominal categories appear in Table A1. The 12 activities were equally distributed (n=4) among the three physical intensity categories of *high*, *medium*, and *low*.

The park benefits for mapping were based on recreation experience items developed by Driver et al. (1991) who identified 19 benefit domains that were reduced to 12 items and used in the Brown et al. (2014) urban park study. These items were as follows: enjoy nature, get exercise/fitness, escape stress, enjoy tranquility, spend time with friends, observe nature, be around good people, do

Activity markers	Physical Intensity Level	Benefit markers	Type
Walking	Moderate	Enjoy nature	Environmental
Running or jogging	High	Get exercise/fitness	Physical
Cycling	High	Escape stress	Psychological
Play sport	Moderate	Enjoy tranquility	Psychological
Resting/sitting	Low	Spend time with friends	Social
Social activities	Low	Observe nature	Environmental
Dog walking	Moderate	Be around good people	Social
Supervise children playing	Low	Do something creative	Psychological
Observe nature/wildlife	Low	Connect with family	Social
Water activities	Moderate	Place to think/reflect	Psychological
Use exercise equipment	High	Rest/relax	Psychological
Boot camp/fitness program	High	Spending time outside	Environmental

something creative, connect with family, place to think/reflect, place to rest/relax, and spending time outside. These benefits were classified into four groups based on the work of Moore and Driver (2005: p. 29): psychological, physical health (a subset of psychophysiological benefits), environmental, and social benefits.

Study participants were requested to identify activities they did in green space over the past two weeks in the Brisbane LGA. Upon marker placement, a pop-up window asked for the frequency and duration of the activity. No time period was specified for the mapping of benefits. To ensure spatial precision in marker placement, markers could only be placed when the Google® maps zoom level was 17 which approximates a 1:4500 map scale. Participants were encouraged to place at least 20 markers (activities + benefits).

Following the mapping activity, participants were redirected to a set of text-based survey questions that collected more information about their greenspace use and sociodemographic information for comparison with census data.

Table A.1 List of markers (icons) for park activities and benefits used in the mapping application. Activity markers were classified into one of three physical intensity levels (Low, Moderate, High) based on assumed MET levels associated with the activity. Park benefits were classified into one of four benefit types (Physical, Environmental, Psychological, and Social).

A.3.4. Data analysis

The spatial data (location and marker type) and non-spatial data (responses to survey questions) were analyzed using ArcGIS® (v10.4) and SPSS® (v24) software. Markers placed outside the study area boundary were excluded from analyses as the focus of this study was park activities and benefits within the Brisbane City Council (BCC) local government area. A total of 8,634 physical activity and benefit markers were available for analyses.

To assess the spatial representativeness of participants within the study area, we compared the proportion of people living in each postcode area using ABS census data (2011) with the proportion of participants in each area. The expected (census) vs. observed (participants) proportions were used to calculate z scores for statistical inference. For example, if a postcode contained 3% of the Brisbane population, and the participant proportion for the postcode was 1%, the postcode would be spatially under-represented. We also assessed spatial representativeness based on the number of points mapped rather than the number of participants. Significant under- or over-represented postcodes were plotted on a map of the study area to indicate potential spatial bias.

To analyze the level of physical activity and types of park benefits occurring within the greater Brisbane area, parks and reserves were classified based on an adapted NRPA park typology (Mertes and Hall, 1996). Table A2 shows the original NPRA classifications and the operational definitions used in this study. Parks were classified into one of eight mutually exclusive categories: (1) Mini-parks consisting of parks/reserves less than 0.4 hectares in size; (2) Neighborhood parks that ranged in size between 0.4 and four hectares; (3) Community parks ranging between 4 and 20 hectares; (4) Large urban parks ranging between 20 and 50 hectares; (5) Schools with greenspaces that are potentially accessible to the public; (6) Sports parks/complexes designed primarily for sporting activities such as football/cricket ovals and that contain relatively little native vegetation; (7) Natural parks that are greater than 50 hectares in size and dominated by native vegetation; (8) Linear parks consisting of parks along the Brisbane River, other creeks and tributaries, and coastal strips. The majority of these linear parks contained connecting trails.

To prepare the data for analysis, physical activity and benefit markers were spatially intersected with park/reserve boundaries, of which 1,133 markers (13%) were located outside formally designated parks/reserves/schools. A total of 9,506 markers (87%) were classified into 845 parks/schools out of 2,350 park/schools in the study boundary area.

Table A.2 Park classification used in this study adapted from NRPA classifications (Mertes and Hall, 1996).

NRPA Classifications	NRPA Size & Location Guidelines	Classification in this study	Operational definition for BCC	Number (%) of activity/ benefit markers[a]	Number of unique units
Mini-park	Mini-park—between 2500 sq. ft. and one acre, less than 1⁄4 mile in residential setting	**Mini-park (1)**	Parks/reserves less than 0.4 hectares	241 (3%)	88
Neighborhood park	Neighborhood—5 to 10 acres optimal, 1⁄4 to 1⁄2 mile distance	**Neighborhood (2)**	Neighbourhood—0.4 to 4 hectares	1162 (13%)	297
Community park	Community—usually between 30 and 50 acres, 1⁄2 to 3 mile distance	**Community (3)**	Community—between 4 and 20 hectares	1836 (21%)	171
Large urban park	Large Urban Park—usually a minimum of 50 acres with 75 or more acres optimal, usually serves entire community	**Large Urban (4)**	Large—between 20 and 50 hectares	598 (7%)	41
School	School-park—variable size, location determined by school	**School (5)**	School grounds—variable in size, identified as educational facility (includes both state and private schools)	109 (1%)	21
Special Use Sports Complex	Special use—size variable, location variable Sports complex—usually a minimum of 25 acres with 40-80 acres optimal, strategically located	**Sports (6)**	Minimum of 10 hectares, dominated by sporting facilities, with little natural vegetation.	80 (1%)	8
Natural Resource Areas	Natural resource areas—size variable, location depends on availability and opportunity	**Natural park (7)**	Natural resource areas—greater than 50 hectares, dominated by natural vegetation	1510 (17%)	35
Park Trails /Connector Trails	Trails--.5 miles per 1000 (1983 NRPA standard), location variable	**Linear park (8)**	Size and location variable; mostly along waterways in BCC	2257 (25%)	184

Associations between physical park activities, park type, and park size

The 12 activity markers were spatially intersected with the parks located in the greater Brisbane area. Activities not falling within any park, reserve, or school boundary were classified as "outside". The activity markers were classified into one of three physical intensity categories based on an assumed MET level: (1) ***low*** intensity activities were associated with sitting, standing, and observing behavior; (2) ***moderate*** intensity activities were associated with walking, water-based activities, or playing sport; (3) ***high*** intensity activities were those associated with running/jogging, cycling, or fitness/boot camp. The park activities were cross-tabulated by park type to generate chi-square statistics and adjusted standardized residuals. Chi-square residuals assess the strength of association between two categorical variables following a statistically significant chi-square result. A standardized residual is the difference between the observed frequency and the expected frequency divided by the standard error of the residual. Standardized residuals provide a normalized score like a *z* score, and if greater than +2.0, indicate significantly more activities than would be expected, while standardized residuals less than -2.0 indicate fewer activities than expected.

To assess the potential relationships between park size, park type, and the physical health benefits associated with park activities, Pearson's product moment correlation was calculated between physical activity scores and park size for each park that contained a minimum of five or more mapped activities. The physical activity score was calculated for each park by summing the products of mapped park activities multiplied by the nominal MET category for the activity. For example, if a park had two resting/sitting activity markers (MET category 1), two walking markers (MET category 2), and one jogging marker (MET category 3), the physical activity score for the park would be (2 x 1 + 2 x 2 + 1 x 3 = 9). The physical activity scores for each park were plotted by park type.

To assess whether the potential influence of park size on mapped activities was significant, we ran a general linear model with the number of mapped activities and the physical activity scores as dependent variables, park type as the independent variable, and park size as a model covariate.

Associations between park benefits, park type, and park size

The 12 park benefit attributes were grouped into four types of benefits: (1) ***physical*** (get exercise/fitness); (2) ***environmental*** (enjoy nature, observe nature, spend time outside); (3) ***psychological*** (escape stress, enjoy tranquility, rest/relax, think/reflect, do something creative; and (4) ***social*** (spend time with friends, be around good people, connect with family) and spatially intersected with parks in study area. Cross-tabulations were generated with the chi-square statistic and standardized residuals to determine significant associations between park type and benefit classifications. The relationship between park size, measured in hectares, and the number of mapped park benefits was analyzed using Pearson's product moment correlation for each park with five or more mapped benefits. The results were graphically plotted by park type.

To assess whether the influence of park size on mapped benefits was significant, we ran a general linear model with the number of mapped benefits as a dependent variable, park type as the independent variable, and park size as a covariate.

Diversity of physical activities and benefits by park type and size

We analysed the diversity of activities and benefits by park type using the Shannon diversity index (Shannon, 1948) for all parks with five or more activities and benefits. The Shannon diversity index accounts for both the abundance and evenness of mapped attributes with index values typically falling within the range of 1.5 to 3.5. Larger index values indicate greater diversity of activities or benefits for a given park. The diversity of park activities and benefits was calculated as follows:

$$-\Sigma p_i \ln p_i$$

where p_i, is the proportional abundance of the *i*th park attribute (activity or benefit) = (n_i/N).

The Shannon index values were calculated for both physical activities and benefits. Spearman rank correlation coefficients were calculated between park size and the diversity indices for all park types combined and for individual park types. A one-way analysis of variance (ANOVA) was performed to determine whether mean diversity indices for activities and benefits differed by park type. Brown (2008) previously found larger urban parks to have a greater diversity of values for urban residents.

Distribution of activities and benefits as a function of distance from domicile

Study participant domicile locations were geocoded from addresses (household sample) or estimated based on the location of the street intersection nearest their home (volunteer sample). The Euclidean distance was calculated in GIS from domicile to each physical activity and benefit. A one-way analysis of variance (ANOVA) was performed to determine the mean distances of mapped activities and benefits to participant domicile and whether these differences were statistically significant.

Spatial distribution of park benefits

To visualize the spatial distribution of park benefits within the Brisbane study area, we categorized each park with two or more mapped benefits (n=355) and classified each according to the most frequently mapped benefit category (physical, environmental, psychological, and social). The parks were symbolized by total number of benefits and benefit type and plotted on a map of the study area using park centroids. To augment visual analysis, we calculated the observed mean distance and the nearest neighbor ratio (R) for each class of parks by benefit category to measure the relative clustering and spatial dispersion of parks.

A.4 RESULTS

A.4.1. Participant characteristics

A total of n=816 study participants mapped one or more spatial attributes in the study resulting in 11,421 mapped attributes, of which 11,187 were located inside the study area and used in subsequent analyses. There were a total n=719 full survey completions where participants mapped locations and answered the text-based survey questions following the mapping activity. Study participants were divided between random household sample respondents (n=541) and volunteer participants (n=275). The response rate for the random household sample was about 8% (541/7096) after accounting for non-deliverable recruitment letters. For the volunteer sample, it is not possible to calculate a traditional response rate. Other internet-based, PPGIS studies of the general public using probability household surveys have reported about a 10% response rate (Pocewicz et al. 2012) or more recently, a 12% response rate using a similar method in Australia that included multiple follow-up reminders (Karimi et al. 2015).

With respect to mapping behavior, the volunteer sampling group mapped significantly more activity and benefit markers on average than the household sampling group (t-tests, $p < 0.05$). For specific activity categories, volunteers mapped significantly more “play sport”, “social activities”, “dog walking”, “observing nature/wildlife”, and “water activities” than the household sample (t-tests, $p < 0.05$). With respect to benefit categories, volunteers mapped significantly more “get exercise/fitness”, “enjoy tranquility/avoid crowds”, “spend time with friends”, “observe/study nature”, “be around good people”, “do something creative”, and “connect with family markers” ($p < 0.05$).

We compared study participant demographic variables with census data from the greater Brisbane area (ABS, 2011) to assess participant representativeness of the Brisbane population (see Table A3). About 49% of participants were female (ABS census=51%) with a median age of 53 (ABS census=35) and an age range of 18 – 87 years. About 43% of participants were in families with children (ABS census=45%). About 68% of participants reported formal education attainment of a Bachelor’s degree or postgraduate education (ABS census=20%) and about 27% reported weekly income of $2,000 or more (ABS census=7%). Thus, the Brisbane participant samples, both random household and volunteer, were biased toward older participants with higher levels of formal education and income than the general Brisbane population. The sampling bias toward older, more highly educated, and higher income levels and is consistent with other reported PPGIS studies (Brown and Kyttä 2014).

Table A.3 Participant profile and statistics

	All	Household	Volunteer
Number of participants (mapped one or more locations)	816	541	275
Number completing post-mapping survey	719	496	223
Number of locations mapped	11,421	6326	5095
Range of locations mapped (minimum/maximum points)	1 - 138	1 - 98	1 - 138
Mean (median) of all markers mapped[1]	14.0 (9.0)	11.7 (8)	18.5 (12)
Mean (median) of activities mapped[1]	5.7 (4.0)	5.0 (3.0)	6.9 (5.0)
Mean (median) of best places mapped[1]	5.4 (2.0)	4.5 (2.0)	7.1 (2.0)
Mean (median) of actions mapped[1]	3.0 (0.0)	2.2 (0.0)	4.5 (1.0)
Knowledge of places (%)			
Excellent	9.3	6.7	15.2
Good	40.9	38.8	45.7
Average	39.6	42.8	32.3
Below average	8.8	10.1	5.8
Poor	1.4	1.6	.9
Years lived in Brisbane (mean)	30.9	32.5	27.5
Gender (ABS, 2011: Male 49.3%)			
Female (%)	48.5	45.1	56.1
Male (%)	51.5	54.9	43.9
Age in years (mean/median) (ABS, 2011: median 35)	52.1 / 53.5	53.9 / 55	48.1 / 47
Education (%) (ABS, 2011: 20.2% Bachelors/postgraduate)			
Less than Bachelors	32	35	26
Bachelor's degree/postgraduate	68	65	74
Income (weekly) (ABS, 2011: 7% $2,000 or more)			
$2,000 or more (%)	27	28	23
Lifecycle (%) (ABS, 2011: 45%)			
Couple family with children	43	45	41
Frequency of park use (%)			
At least once per week	78	75	85
At least once per fortnight	9	10	8
At least once per month	5	6	3
Less than once per month	8	9	4

[1] Mean differences in the number of markers mapped by household and volunteer groups are statistically significant (t-tests, $p < 0.05$).

From the survey questions, study participants have lived in the Brisbane area for an average of 31 years. Over 50% of participants rated their knowledge of Brisbane parks/reserves and other greenspaces as "excellent" or "good" with about 40% rating their knowledge as "average". Less than 2% rated their knowledge as "poor". In terms of park/reserve use frequency, about 78% of participants use parks at least once a week with another 9% using the parks at least once every two weeks or once a month (5%).

The spatial representativeness of participants were assessed by comparing the proportion of participants by postcode with the proportion of Brisbane residents living in the postcode as reported in census data. Significant deviations in postcode proportions with z scores greater than +2.0 or less than -2.0 were plotted on a map (see Figure A1). There was some spatial bias toward greater participation in four postcodes (indicated in green), and disproportionately less participation in one postcode area (indicated in red). Analysis based on the proportion of total activity and benefit points mapped rather than the number of participants indicated that three postcodes were over-represented. Thus, spatial bias in response was relatively low with most study participants spatially distributed across the study area in rough proportion to the overall population.

A.4.2. Relationships between physical activities, park type, and park size

There was a statistically significant association between physical activity markers (coded as *low*, *moderate*, and *high* MET intensity) and park type for all respondents (X^2=82.9, df=16, $p < 0.001$) and for the household (X^2=38.8, df=16, $p < 0.001$) and volunteer (X^2=58.5, df=16, $p < 0.001$) samples respectively (Table A4). The largest number of *high* MET activities were associated with linear parks for all sampling groups, followed by community parks. The proportion of *high* MET activities was also significantly larger than expected outside formal park boundaries (residuals greater than +2.0), a logical result given that high MET activities such as jogging and cycling often include geographic areas outside of park boundaries as part of the activity. The smaller urban park classes—mini-park and neighborhood—contained more *low* MET activities and fewer *high* MET activities than would be expected based on chi-square residual values.

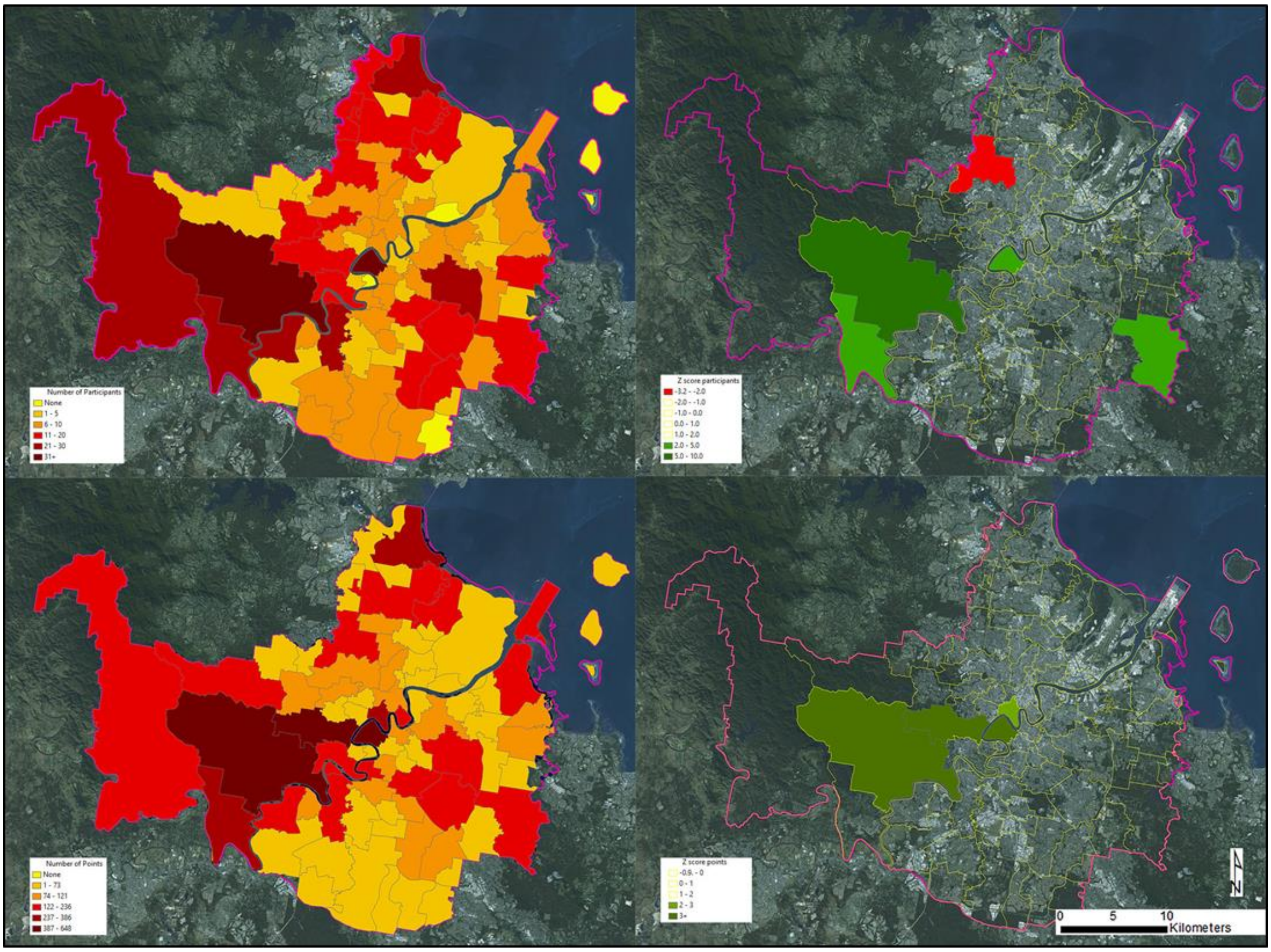


Figure A.1 Distribution of (a) number of participants and (b) mapped points (activities and benefits) by postcode area in Brisbane. Z scores indicate whether number of participants (c) and points (d) are significantly greater (green) or less than expected (red) based on population proportions.

The relationship between physical activities, park type, and park size was further examined by plotting aggregated physical activity scores by park type and size for parks with more than five mapped activities (Figure A2). The bivariate correlation between activity score and park size was significant, but moderate in strength ($r=0.41$, $p < 0.05$) suggesting larger parks provide more opportunities for physical activities and associated health benefits. When park size was treated as a covariate in a general linear model (GLM) with aggregated activity score as the dependent variable and park type as the independent variable for parks with more than five mapped activities ($n=216$), the model was significant ($F=3.5$, $p < 0.001$) but weak ($R^2= 0.11$). The park size covariate was not significant in the model ($p > 0.05$). Natural parks had the largest mean activity scores, followed by linear parks, and then large urban parks. The lowest mean activity scores were found in mini-parks. When the model was run on the number of activities as the dependent variable rather than the aggregated MET activity score, and all parks were included in the analysis regardless of the number of activity markers (n=755), the model was significant ($F=17.4$, $p < 0.001$, $R^2=.140$) with park size being a significant covariate ($p < 0.05$). Thus, the number of physical activities mapped appears significantly related to park size, with fewer activity markers, on average, being placed in the large number of mini- and neighborhood parks across Brisbane.

A.4.3. Relationships between park benefits, park type, and park size

There were statistically significant associations between benefit markers and park type for all respondents ($X^2=120.1$, $df=24$, $p < 0.001$) and for the household ($X^2=85.3$, $df=24$, $p < 0.001$) and volunteer ($X^2=75.1$, $df=24$, $p < 0.001$) samples respectively (Table A5). Environmental benefits were over-represented in natural parks while physical benefits were over-represented in linear parks as indicated by residuals greater than +2.0. Environmental benefits were under-represented in linear parks and social benefits were under-represented in natural parks (residuals < -2.0). Community parks were over-represented with social benefits.

The relationship between park benefits, park type, and park size was further examined by plotting the number of benefits by park type and size for parks with five or more mapped benefits (Figure A3). Natural parks had the largest mean number of mapped benefits, followed by large urban parks, and community parks. The lowest mean number of benefits was found in sports parks. The bivariate correlation between the number of mapped benefits and park size was significant, but moderate in strength ($r=0.52$, $p < 0.05$). When park size was treated as a covariate in a general linear model (GLM) with the number of benefits as the dependent variable and park type as the independent variable for parks with five or more mapped benefits (n=176), the model was significant ($F=4.5$, $p < 0.000$, $R^2=0.16$). The park size covariate was significant ($p =0.079$) at the 0.10 level of significance in the model.

Table A.4 Cross-tabulation of physical activity level by park type showing the number and percentage of activity markers with adjusted standardized chi-square residuals for all participants and for two sampling groups (random household and volunteer). Adjusted standardized residuals +2.0 or greater (green) indicate more activity markers than expected and standardized residuals -2.0 (pink) or less indicate fewer markers than expected.

	Physical activity level (all respondents)[a]				**Physical activity level (Household)[b]**				**Physical activity level (Volunteer)[c]**			
Park Type	Low	Moderate	High	Total	Low	Moderate	High	Total	Low	Moderate	High	Total
Outside of park	130	291	165	586	74	218	104	396	56	73	61	190
	10.4%	14.2%	15.6%	13.4%	11.6%	15.7%	19.2%	15.4%	9.2%	11.0%	11.8%	10.6%
	-3.7	1.3	2.4		-3.1	.4	2.7		-1.4	.4	1.1	
Mini-park	53	66	20	139	30	50	11	91	23	16	9	48
	4.2%	3.2%	1.9%	3.2%	4.7%	3.6%	2.0%	3.5%	3.8%	2.4%	1.7%	2.7%
	2.5	.1	-2.8		1.8	.2	-2.1		2.0	-.5	-1.6	
Neighborhood park	231	322	103	656	114	217	58	389	117	105	45	267
	18.5%	15.7%	9.7%	15.1%	17.9%	15.6%	10.7%	15.2%	19.1%	15.8%	8.7%	14.9%
	4.0	1.1	-5.5		2.2	.7	-3.3		3.6	.8	-4.7	
Community park	274	427	183	884	143	292	102	537	131	135	81	347
	21.9%	20.8%	17.3%	20.3%	22.4%	21.0%	18.8%	20.9%	21.4%	20.4%	15.7%	19.4%
	1.7	.8	-2.8		1.1	.2	-1.4		1.6	.8	-2.5	
Large urban park	65	117	60	242	32	75	27	134	33	42	33	108
	5.2%	5.7%	5.7%	5.6%	5.0%	5.4%	5.0%	5.2%	5.4%	6.3%	6.4%	6.0%
	-.6	.4	.2		-.3	.5	-.3		-.8	.4	.4	
Schools	20	31	18	69	5	14	3	22	15	17	15	47
	1.6%	1.5%	1.7%	1.6%	0.8%	1.0%	0.6%	0.9%	2.5%	2.6%	2.9%	2.6%
	.1	-.4	.4		-.2	.9	-.9		-.3	-.1	.5	
Sports park	13	27	10	50	9	16	9	34	4	11	1	16
	1.0%	1.3%	0.9%	1.1%	1.4%	1.2%	1.7%	1.3%	0.7%	1.7%	0.2%	0.9%
	-.4	1.0	-.7		.2	-.8	.8		-.8	2.6	-2.0	
Natural park	152	248	144	544	74	175	63	312	78	73	81	232
	12.2%	12.1%	13.6%	12.5%	11.6%	12.6%	11.6%	12.2%	12.7%	11.0%	15.7%	13.0%
	-.4	-.7	1.3		-.5	.8	-.4		-.2	-1.9	2.2	
Linear park	311	522	354	1187	156	331	165	652	155	191	189	535
	24.9%	25.5%	33.5%	27.2%	24.5%	23.8%	30.4%	25.4%	25.3%	28.8%	36.7%	29.9%
	-2.2	-2.5	5.2		-.6	-2.0	3.0		-3.0	-.8	4.0	
Total markers	1249	2051	1057	4357	637	1388	542	2567	612	663	515	1790
	28.7%	47.1%	24.3%		24.8%	54.1%	21.1%		34.2%	37.0%	28.8%	

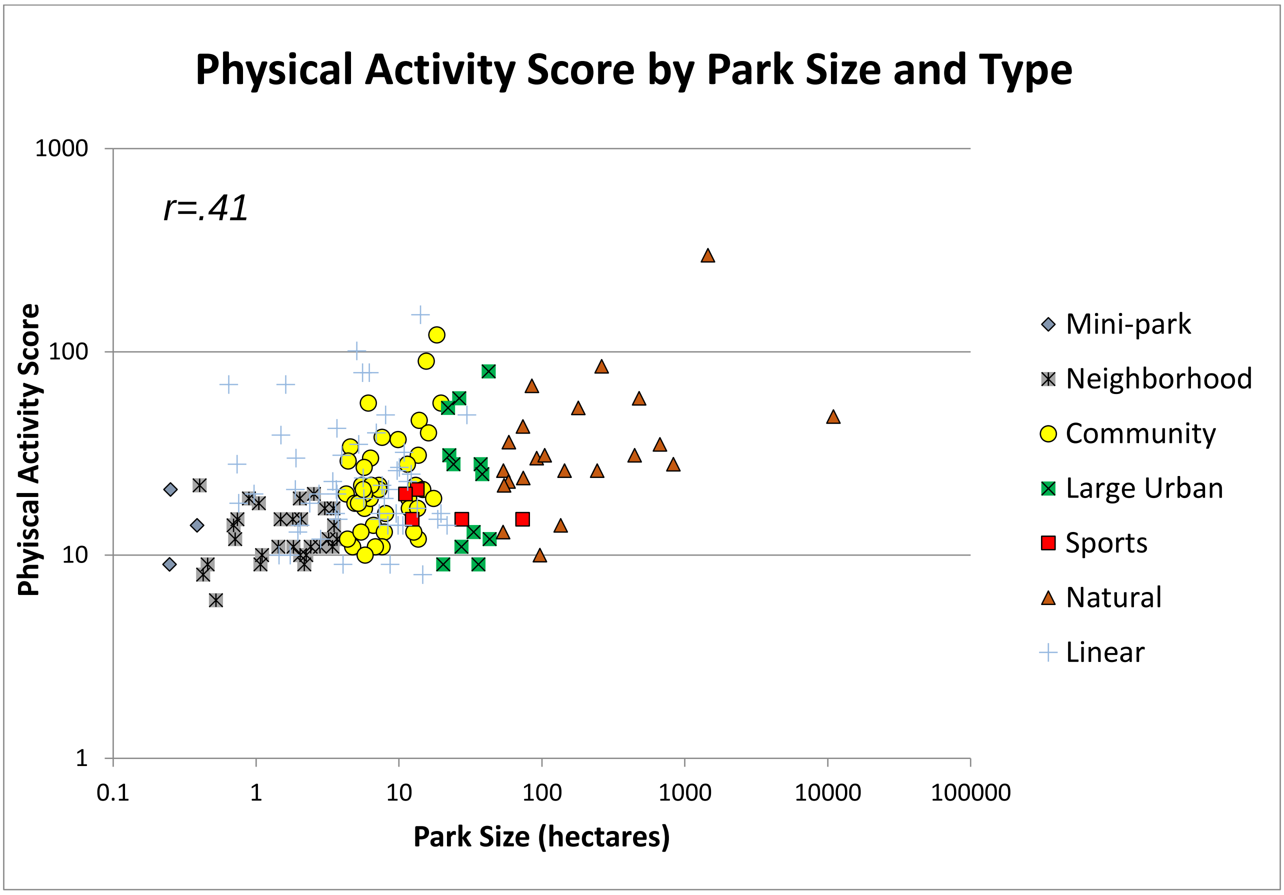


Figure A.2 Relationship between aggregated physical activity scores by park type and park size (hectares). Each activity was multiplied by associated MET intensity level category (low=1, moderate=2, high=3) to calculate physical activity score. Parks with greater than five mapped activities were used in the calculation.

Table A.5 Cross-tabulation of park benefit by park type showing the number and percentage of benefit markers with adjusted standardized chi-square residuals for all responses and two sampling groups (random household and volunteer). Adjusted standardized residuals +2.0 or greater (green) indicate more benefit markers than expected and standardized residuals -2.0 (pink) or less indicate fewer markers than expected.

	Benefit category (all respondents)[a]					Benefit category (Household)[b]					Benefit category (Volunteer)[c]				
Park Type	Phys	Environ	Psych	Social	Total	Phys	Environ	Psych	Social	Total	Phys	Environ	Psych	Social	Total
Outside of park	99	164	135	89	487	78	111	84	61	334	21	53	51	28	153
	12.8%	10.4%	11.1%	12.7%	11.4%	17.0%	12.8%	12.4%	17.0%	14.1%	6.6%	7.4%	9.4%	8.2%	8.0%
	1.3	-1.6	-.4	1.2		2.0	-1.4	-1.6	1.7		-1.0	-.7	1.4	.1	
Mini-park	14	37	27	12	90	12	27	20	10	69	2	10	7	2	21
	1.8%	2.3%	2.2%	1.7%	2.1%	2.6%	3.1%	2.9%	2.8%	2.9%	0.6%	1.4%	1.3%	0.6%	1.1%
	-.6	.8	.3	-.8		-.4	.4	.0	-.2		-.9	1.0	.5	-1.0	
Neighborhood park	75	145	155	75	450	54	87	99	28	268	21	58	56	47	182
	9.7%	9.2%	12.7%	10.7%	10.5%	11.8%	10.1%	14.6%	7.8%	11.3%	6.6%	8.1%	10.3%	13.7%	9.5%
	-.9	-2.2	2.9	.2		.3	-1.5	3.1	-2.3		-1.9	-1.6	.8	2.9	
Community park	136	316	273	180	905	78	171	138	88	475	58	145	135	92	430
	17.5%	20.0%	22.4%	25.7%	21.2%	17.0%	19.8%	20.3%	24.5%	20.1%	18.3%	20.3%	24.9%	26.9%	22.5%
	-2.7	-1.4	1.2	3.2		-1.9	-.3	.2	2.3		-1.9	-1.7	1.6	2.2	
Large urban park	56	137	91	54	338	22	68	50	27	167	34	69	41	27	171
	7.2%	8.7%	7.5%	7.7%	7.9%	4.8%	7.9%	7.4%	7.5%	7.1%	10.7%	9.7%	7.6%	7.9%	8.9%
	-.8	1.4	-.7	-.2		-2.1	1.1	.4	.4		1.2	.9	-1.3	-.7	
Schools	10	12	10	5	37	5	3	1	3	12	5	9	9	2	25
	1.3%	0.8%	0.8%	0.7%	0.9%	1.1%	0.3%	0.1%	0.8%	0.5%	1.6%	1.3%	1.7%	0.6%	1.3%
	1.4	-.6	-.2	-.5		2.0	-.8	-1.6	.9		.5	-.1	.9	-1.3	
Sports park	6	9	5	7	27	6	6	5	6	23	0	3	0	1	4
	0.8%	0.6%	0.4%	1.0%	0.6%	1.3%	0.7%	0.7%	1.7%	1.0%	0.0%	0.4%	0.0%	0.3%	0.2%
	.6	-.4	-1.2	1.3		.8	-1.1	-.7	1.5		-.9	1.6	-1.3	.4	
Natural park	168	443	246	79	936	89	224	132	33	478	79	219	114	46	458
	21.6%	28.1%	20.1%	11.3%	21.9%	19.4%	25.9%	19.4%	9.2%	20.2%	24.9%	30.7%	21.0%	13.5%	23.9%
	-.2	7.5	-1.7	-7.4		-.5	5.2	-.6	-5.7		.5	5.3	-1.9	-5.0	
Linear park	212	316	279	200	1007	115	168	150	103	536	97	148	129	97	471
	27.3%	20.0%	22.9%	28.5%	23.5%	25.1%	19.4%	22.1%	28.7%	22.7%	30.6%	20.7%	23.8%	28.4%	24.6%
	2.7	-4.2	-.7	3.4		1.3	-2.9	-.4	2.9		2.7	-3.0	-.5	1.8	
Total markers	776	1579	1221	701	4277	459	865	679	359	2362	317	714	542	342	1915
	18.1%	36.9%	28.5%	16.4%		19.4%	36.6%	28.7%	15.2%		16.6%	37.3%	28.3%	17.9%	

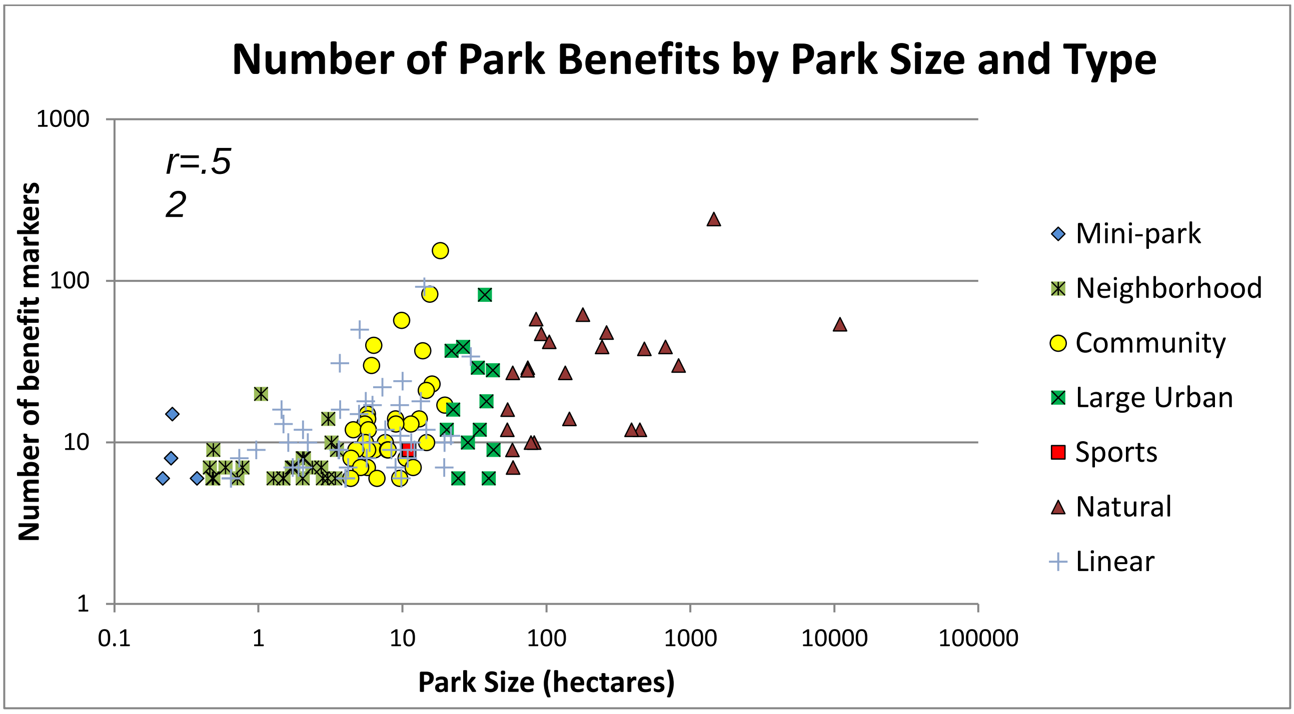


Figure A.3 Relationship between aggregated benefits by park type and park size (hectares). Parks with greater than five mapped benefits were used in the calculation.

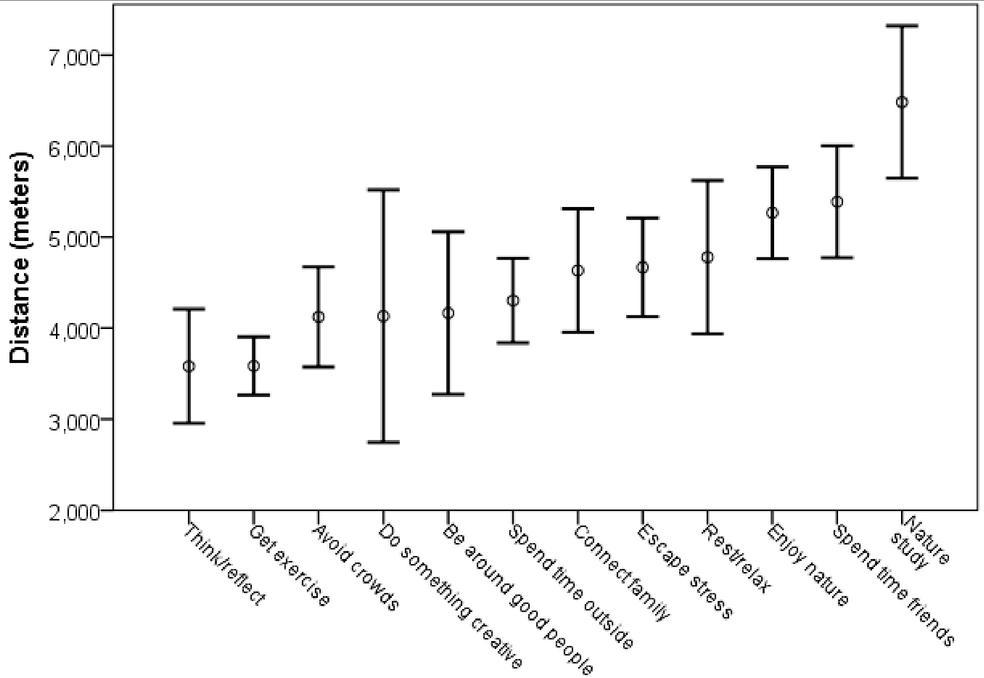
A.4.4. Diversity of activities and benefits by park type and size

We examined the diversity of activities and benefits by park type using the Shannon diversity index. For all park types combined, there were significant bivariate rank correlations between the diversity of activities and park size (r=0.56, p < 0.001) and diversity of benefits and park size (r=0.49, p < 0.001). Within a specific park type, there were significant correlations with park size between activity diversity and community parks (r=0.43, p < 0.01), large urban parks (r=0.78, p < 0.001), sports parks (r=0.97, p < 0.01), natural parks (r=0.58, p < 0.01), and linear parks (r=0.60, p < 0.001). Benefit diversity was significantly correlated with park size for community parks (r=0.59, p < 0.001), large urban parks (r=0.55, p < 0.05), and linear parks (r=0.30, p < 0.05).

We used ANOVA with Tukey HSD post-hoc tests to examine pairwise comparisons of mean activity diversity by park type. Neighborhood park activity diversity was significantly lower than all other park types (p < 0.05), with all other park types being similar in mean diversity (p > 0.05). For benefit diversity, neighborhood park diversity was significantly lower than all other park types (p < 0.05) and natural park diversity was significantly higher than all other park types (p < 0.05). Mean benefit diversity was similar for community, large urban, and linear parks.

A.4.5. Distribution of activities and benefits as a function of distance from domicile

We examined the distribution of activities and benefits as a function of distance from domicile. Mean distances were calculated from domicile to each type of mapped activity or benefit and an ANOVA model was used to assess whether mean distances from domicile varied by activity or benefit type. With respect to activities, the shortest mean distance was for using exercise equipment (1827 m) while the longest distance was for social activities (4811 m). An error plot for distances between domicile and all mapped activities appears in Figure A4 with statistically significant differences indicated in the table below the plot (ANOVA, $p < 0.05$, Tukey HSD). For benefits, the shortest mean distance was for places to think/reflect (3582 m) and to get exercise (3586 m) and the longest distances was for nature study (6482 m) and spending time with friends (5389 m). The mean distances to benefits were logically consistent with mean distances to activities associated with the benefits. Specifically, the activities and benefits of getting exercise was closest to domicile while the activities and benefits associated with nature and social activities were most distant from domicile.



	Enjoy nature	Get exercise	Escape stress	Avoid crowds	Spend time friends	Nature study	Be around good people	Do something creative	Connect with family	Think/ reflect	Rest/ relax	Spend time outside
Enjoy nature												
Get exercise	*											
Escape stress												
Avoid crowds												
Spend time friends		*										
Nature study		*	*	*								
Be around good people						*						
Do something creative												
Connect with family						*						
Think/reflect	*				*	*						
Rest/relax						*						
Spend time outside						*						

Figure A.4 Error bar plot showing mean distance (meters) and 95 percent confidence intervals for 12 benefits from study participant domicile to mapped location with table showing benefit distances that are significantly different (ANOVA, Tukey HSD, $p < 0.05$)

A.4.6. Spatial distribution of park benefits in the study area

Each park with two or more mapped benefits (n=355) was classified according to the most frequently mapped benefit type (physical, environmental, psychological, and social), was plotted on a map, and nearest neighbor statistics were calculated. If there was a tie in the most frequently mapped benefit type, the park was classified by both benefit types. The most frequent park class by benefit type was "environmental" (n=210) with a mean nearest neighbor of 1084 meters and a nearest neighbor ratio of 0.81 ($z=-5.36$, $p < 0.001$). The least frequent park class by benefit was "social" (n=22) with a mean nearest neighbor of 3340 meters and a nearest neighbor ratio of 1.32 ($z=2.85$, $p < 0.01$). Parks where physical benefits were most frequent (n=48) had a mean nearest neighbor of 1524 meters and a nearest neighbor ratio of 0.84 ($z=-2.17$, $p < 0.05$) while parks where psychological benefits were most frequent (n=75) had a mean nearest neighbor of 1566 meters and a nearest neighbor ratio of 0.85 ($z=-2.49$, $p < 0.05$). Visually, these results are shown in Figure A5 with fewer and more spatially dispersed "social" parks (red) and a greater number and more clustered "environmental" parks (green). Parks where "psychological" benefits were most frequent (blue) were most proximate to the Brisbane central business district (CBD) while parks where "environmental" benefits were the most frequent type (green) are evident on the periphery of the Brisbane study area and coincide with natural forest parks in the western and northern reaches of the Brisbane urban area.

A.5 DISCUSSION

In this study, we evaluated the use of public participation GIS (PPGIS) methods to assess park benefits for a large urban park system (Brisbane, Australia). Previous research used participatory mapping methods to assess park benefits for a suburb located within the larger urban area of Adelaide, Australia (Brown et al. 2014). The scaling-up of the research to a large urban park system necessarily involved changes in research design and implementation with the potential to influence research outcomes. In addition to validating previous findings on the public benefits of different urban park types, we reflect on the challenges of scaling-up of participatory mapping research methods for a large and diverse urban park system.

A.5.1. Urban park classification and urban planning

One of the greatest challenges—and arguably—one of the most important with implications for both public benefit analysis and urban planning is the park classification system that describes the structure of urban park system (size, components, and spatial configuration). Classification systems have been guided by physical properties, park features, and the surrounding environment, an approach that is consistent with a planning standards approach to urban planning and design. However, an argument can be made that the provision of urban parks and greenspaces should also be equally informed by an understanding of the distribution of benefits provided by urban parks and greenspaces. The physical presence of parks and greenspaces does not guarantee that the imputed human benefits of parks are actually realized, nor equitably distributed, especially when park access is multi-dimensional with geographic proximity being just one factor among others (Wang et al. 2015b). Further, simply knowing the physical structure of an urban park system does not provide sufficient information for benefit trade-off analysis in decisions regarding the allocation of scarce urban space.

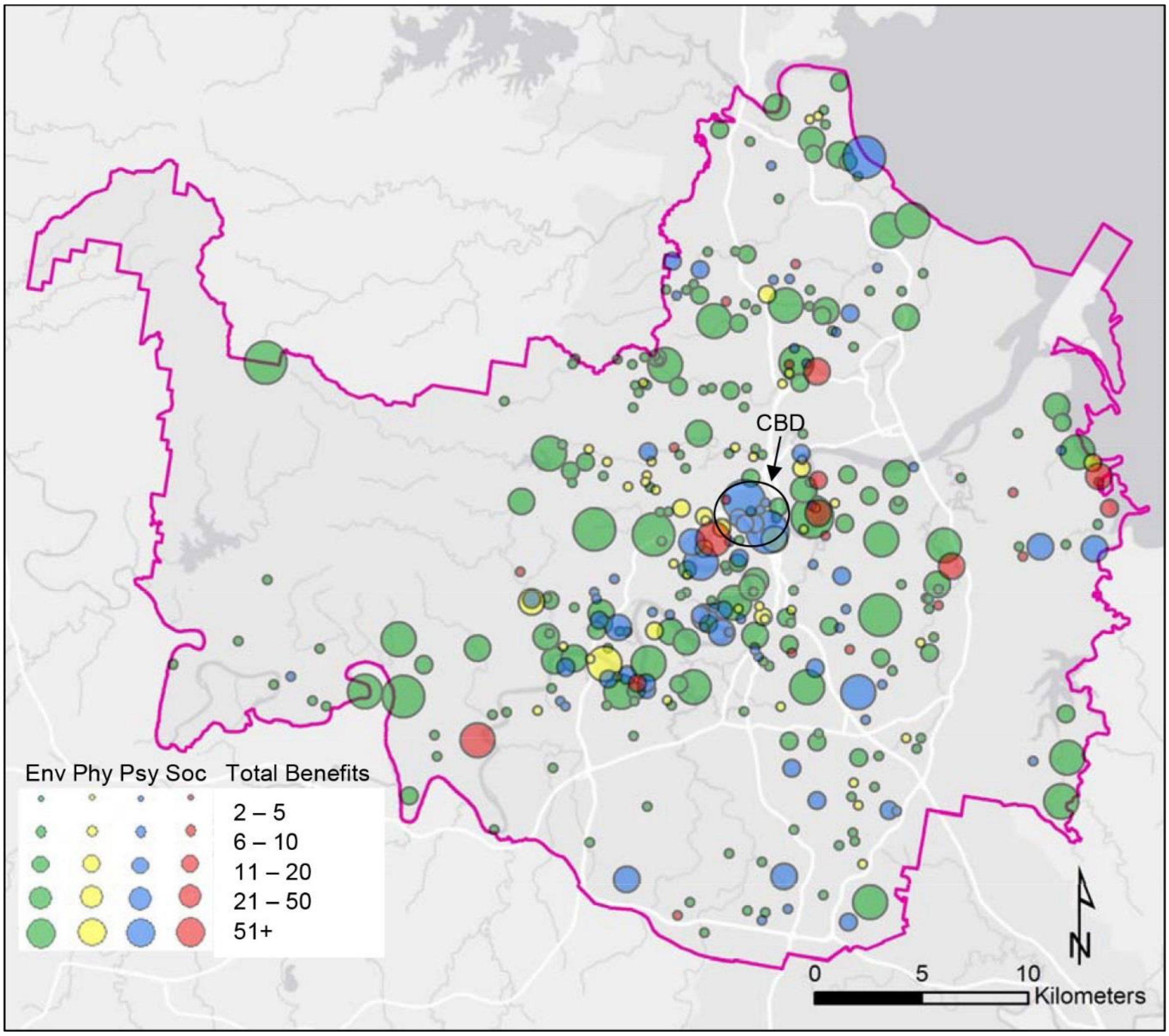


Figure A.5 Map showing the spatial distribution of benefits mapped in parks (displayed as centroids) with two or more mapped benefits. Colors show the most frequent benefit type in the park where Env=environmental, Phy=physical, Psy=psychological, Soc=social. CBD =Central Business District.

The Adelaide suburb research operationalized six classifications from the NRPA park typology (Mertes and Hall, 1996). The NRPA classification system uses the criteria of size, proximity, and function. For example, park types are classified primarily by their size, but some park classes also include proximity to residential areas as a criterion. Sports and recreation parks are identified by function to meet the requirements of the sporting/recreation activity (e.g., soccer fields). This Brisbane study also used the NRPA classification system as a foundation for identifying eight types of urban parks (including schools) primarily based on size, but also included other criteria such as physical shape, waterway contiguity, dominant park function, and the extent of native vegetation. Classifying sports parks in Brisbane posed a challenge because these parks may include other park features (e.g., natural areas) not associated with the sporting activity. Linear parks in this study were classified primarily based on their shape (i.e., elongated and narrow), but with additional consideration for contiguity with physical features such as waterways and the presence of connecting trails. The distinction between sports parks, large urban parks, and natural parks which overlapped in size required a subjective judgement about the dominant function of the park, combined with the extent of native vegetation. In short, classification of parks required some subjective analyst judgement when applying multiple criteria.

In scaling the research to a large urban park system that included over 2,300 designated parks and reserves, we used objective GIS criteria to generate initial park classes, which were then visually assessed for possible reclassification. In our classification system, the park size break points that distinguish neighborhood from community parks lack definitive supporting rationale and empirically, the results were similar for these types of parks. Additional greenspace classification criteria such as those described by Kimpton (2017) that account for the presence and abundance of amenities such as facilities could augment the classification system, as can classification systems that account for additional variables such as land cover, built context, and social context (Ibes 2015).

Historically, the planning for urban parks and greenspaces, to the extent that it has been intentional and proactive, has followed a standards approach based on ratios such as the amount of parkland per population. An enhanced standards approach, as found in the NRPA guidelines (Mertes and Hall, 1996), treats urban parks and greenspaces as a system and assumes that different types of urban parks and greenspaces provide differential human benefits within the system. Our mapping results provide empirical evidence that the systems approach to park classification embodied in the NRPA framework appears sound, even when applied to a large, complex urban park system such as

Brisbane that is characterized by a high level of park diversity. The participatory mapping methods described in this study also assume a systems approach to understanding urban park benefits. The pairing of these two systems approaches (physical structure and social benefit structure) provides an evidence-base to inform future urban park planning. For example, in the Brisbane system, increasing physical health benefits would suggest investment in more linear parks (or greater trail connectivity in existing linear parks), increasing social benefits would suggest investment in community parks, and providing greater psychological benefit would suggest greater investment in neighborhood parks. The environmental benefits of parks and greenspaces already appear ubiquitous across the city

A.5.2. Association of activities and benefits by park type, size, and distance

Consistent with previous research, we found that linear parks, in particular, provide significant health benefits because they provide opportunities to engage in higher intensity aerobic physical activities such as walking, running, and cycling. Given the nature of these activities, these were also mapped disproportionately outside formal parks and reserves. Linear parks play a significant role in facilitating these activities through trails that make these activities safer and more enjoyable. Our linear park results were not as strong as the Adelaide suburb research because Brisbane contains many more parks that were classified as linear based on shape and adjacency to waterways, but some of these parks lack developed trails that make them attractive for walking, running, or cycling longer distances.

The distribution of non-physical park benefits (psychological, environmental, and social) by type of park/reserve was also consistent with previous research. As a system, urban parks provide a full range of public benefits but the benefits appear differentially important based on park type. Natural parks provide disproportionately more environmental benefits while community parks provide disproportionately more social benefits. In this study, neighborhood parks emerged as providing disproportionately more psychological benefits (e.g., escape stress, rest/relax), a benefit/type association that was not significant in the previous study at the suburb scale.

Brown (2008) posited that the diversity of values people hold for parks increases with park size and the proximity of parks to denser urban populations. The Adelaide suburb-level study provided significant evidence for the importance of park size and park type to both physical activity and benefit diversity. In this study, park size and park type were also significantly related to activity diversity and benefit diversity, thus confirming the influence of park type and size when scaled-up to an urban park system with more parks and greater park variability. As a general principle, larger parks provide greater activity and benefit diversity. The diversity of park activities and benefits appear lower for parks such as neighborhood parks, and higher for natural parks.

With respect to distance analyses of activities and benefits to participant domicile, these study results were consistent with the Adelaide suburb study. Physical benefits were located most proximate to participant domicile while social benefits were more distant. Environmental benefits, primarily associated with natural parks, were located most distant from participant domicile which appears logical given the configuration of the park system in Brisbane where larger natural parks are located on the urban periphery. Lin et al. (2014) suggested that the motivation to visit parks and interact with nature in Brisbane is driven more by nature orientation—the affective, cognitive, and experiential relationship individuals have with the natural world—than the availability and proximity of parks. Our study did not measure affinity for nature so we cannot directly assess park use motivation on this variable. However, the opportunity for environmental benefits from parks and greenspaces does not appear to be a limiting factor as parks that provide environmental benefits are spatially distributed throughout the greater Brisbane area (Figure 6).

Our results indicate that Brisbane park users do differentiate park benefits spatially based on park distance from domicile and appear willing to travel longer distances to obtain social and environmental benefits of urban parks in particular. However, the evidence for the importance of distance from domicile as a factor in explaining actual park use and associated benefits appears weak. For example, Schipperijn et al. (2010) did not find distance to greenspaces to be a limiting factor for the majority of the Danish population in explaining the frequency of greenspace use. In the U.S., distance to the closest park was not significantly related to either park use or park physical activity (Kaczynski et al. 2014). In Melbourne, Australia, proximity was not associated with walking to or within public open-spaces (Koohsari et al. 2013). Rather than proximity or geographic access, perceived park access—a multi-dimensional construct—appears to be a stronger predictor of park use in Brisbane and thus the range of benefits associated with urban parks (Wang et al. 2015a).

A.5.3. Research design and validation

Participatory mapping methods can be effectively implemented across large urban areas as demonstrated in this study and other cities such as Helsinki (Kahila-Tani et al. 2016). But given the human diversity and physical heterogeneity of urban areas, ensuring the representativeness of participants (both demographic and spatial) poses one of the greatest challenges to research validity when assessing public benefits from urban parks/reserves. Household surveys are experiencing higher refusal rates where nonresponse is more likely to induce bias in survey estimates (Groves 2006). Our household response rate was low, but consistent with other participatory mapping studies (see Brown 2017). In this study, random household, probability-based sampling was supplemented by a volunteer sample recruited through newsletters, social media, and participant referrals. These recruitment methods achieved acceptable spatial representation across the study area (Figure 1), but probability-sampled participants were demographically biased toward older, more formally educated, and higher income individuals. These demographic results are consistent with findings of a previous survey of Brisbane park users which found park users to be somewhat older and with a higher level of formal education than non-park users (Lin et al. 2014). Our study participants also appeared to be more frequent users of parks than would otherwise be expected. About 78% of participants reported using parks at least once a week compared to about 60 percent found in a previous study (Lin et al. 2014). The participant bias toward more formal education, more familiarity with parks, and more frequent park use was greater in the volunteer sample than the household sample, an expected finding given the presumed greater saliency of parks issues to the volunteer group. A limitation of this study was the under-representation of Brisbane participants by lower socio-economic status or ethnicity, variables that can significantly influence park use and/or behaviour (Dwyer and Gobster 1992; Gobster 2002; Shackleton and Blair 2013). Further, our sampling methods did not directly target children, a key demographic for community health assessment. Participatory mapping methods can be implemented to identify children's behavior (Kyttä et al. 2012) related to park use.

In participatory mapping with a typology of pre-defined attributes, the number of attributes to be mapped are necessarily constrained given the limited time participants are willing to engage in mapping activity. Our list of physical activities to be mapped included several new activities (dog walking, water-based activities, and supervising children in parks) not previously used, but as a web-design trade-off, the list of markers did not provide different MET intensity levels for walking, running, cycling, and sport activities as used in the Adelaide suburb study. In our analyses, we made assumptions about the MET intensity levels for all mapped activities (low, moderate, high) which are open to critique given participant variability in the actual physical intensity of these activities. Nonetheless, our findings regarding physical health benefits by park type based on assumed MET levels were consistent with previous research showing greater physical health benefits with larger urban parks in general, and linear parks in particular.

In the web-based mapping design, the placement of an activity marker was followed by two questions asking about how many times the activity was done in the past two weeks and the aggregate time spent doing the activity over the past two weeks. The purpose of these questions was to better estimate the physical health benefits associated with the mapped activities similar to research using activity-log methods. However, there were data quality issues with greater activity frequencies reported than the presumed maximum of 14 times over the two week period. We removed markers with inconsistencies in the frequency data and ran the analyses by weighting the markers by frequency under the assumption that the activity marker represented multiple visits. The net effect was to weaken the significant associations by park type, a likely result of introducing greater individual variability in park use that masked more fundamental activity/park associations.

The activity duration question asked for responses in hours over the two week period, but many responses appeared to be recorded in minutes. This question had the greatest potential to calibrate the MET data but the data were too inconsistent. In the future, the application would benefit from data error-checking logic to preclude participants from entering obvious out-of-range data. However, even if data quality were higher, large-scale participatory mapping across an urban park system does not appear to be the most appropriate method for achieving accurate physical health data on an individual person or park basis. If an important research objective is to achieve more accurate recording of park activities, physical activity logs or direct observation methods such as SOPARC could be used in combination with participatory mapping to calibrate the results.

A.6 CONCLUSION

In this study, we evaluated participatory mapping methods for assessing urban park benefits. The scaling-up of these methods from the suburb-level to a large urban-park system introduced greater variability in the results but multiple urban park benefits by park type associations were confirmed at the larger urban scale. Participatory mapping, with a focus on the distribution of park benefits in addition to physical design standards, can provide supplemental information to refine and adjust physical park standards.

There is contemporary academic interest in the assessment and analysis of urban areas for ecosystem services (e.g., Gómez-Baggethun and Barton 2013; Rall et al. 2017; Woodruff and BenDor 2016). The participatory mapping methods described in this study provide a means to assess cultural ecosystem services associated with urban parks and greenspaces. However, as noted by Ahearn et al. (2014), the assessment of urban ecosystem services alone does not provide the innovation required to inform routine urban and infrastructure development activity (Ahern et al. 2014). And yet, participatory mapping offers the potential to better inform urban green infrastructure because of its spatially-explicit, systems approach to assessment focused on a range of benefits. Future research could analyze the spatial distribution of park benefits by suburb or neighborhood (spatial disaggregation) to identify social inequities in park benefits that could be addressed through further development of green infrastructure.

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APPENDIX B: HUMAN ETHICS COMMITTEE APPROVAL LETTER

THE UNIVERSITY OF QUEENSLAND

Institutional Human Research Ethics Approval

Project Title:	Brisbane Parks and Green Space Study
Chief Investigator:	A/Prof Jonathan Rhodes
Supervisor:	None
Co-Investigator(s):	Marie Dade, Assoc/Prof Greg Brown
School(s):	School of Geography, Planning and Environmental Management
Approval Number:	2016001148
Granting Agency/Degree:	ARC Discovery Projects DP130100218
Duration:	31st August 2020

Comments/Conditions:

Expedited Review - Low Risk

Note: If this approval is for amendments to an already approved protocol for which a UQ Clinical Trials Protection/Insurance Form was originally submitted, then the researchers must directly notify the UQ Insurance Office of any changes to that Form and Participant Information Sheets & Consent Forms as a result of the amendments, before action.

Name of responsible Committee:
University of Queensland Human Research Ethics Committee A
This project complies with the provisions contained in the *National Statement on Ethical Conduct in Human Research* and complies with the regulations governing experimentation on humans.

Name of Ethics Committee representative:
Professor Emerita Gina Geffen
Chairperson
University of Queensland Human Research Ethics Committee A
Registration: EC00456

Signature G. Geffen Date 12/09/2016

APPENDIX C: SUPPLEMENTARY MATERIAL FOR CHAPTER 2

Table C.1 The final set of papers selected for the literature review database.

	Papers included in the literature review
1	Ai, J.Y., Sun, X., Feng, L., Li, Y.F., Zhu, X.D. 2015. Analyzing the spatial patterns and drivers of ecosystem services in rapidly urbanizing Taihu Lake Basin of China. *Frontiers of Earth Science,* 9: 531-545.
2	Alarcon, G.G., Ayanu, Y., Fantini, A.C., Farley, J., Schmitt, A., Koellner, T. 2015. Weakening the Brazilian legislation for forest conservation has severe impacts for ecosystem services in the Atlantic Southern Forest. *Land Use Policy*, 47: 1-11.
3	Albizua, A., Williams, A., Hedlund, K., Pascual, U. (2015) Crop rotations including ley and manure can promote ecosystem services in conventional farming systems. *Applied Soil Ecology*, 95: 54-61.
4	Allen, K.E. 2015. Trade-offs in nature tourism: contrasting parcel-level decisions with landscape conservation planning. *Ecology and Society*, 20.
5	Anderson-Teixeira, K.J., Duval, B.D., Long, S.P., DeLucia, E.H. 2012. Biofuels on the landscape: Is "land sharing" preferable to "land sparing"? *Ecological Applications*, 22: 2035-2048.
6	Anderson, B.J., Armsworth, P.R., Eigenbrod, F., Thomas, C.D., Gillings, S., Heinemeyer, A., Roy, D.B., Gaston, K.J. 2009. Spatial covariance between biodiversity and other ecosystem service priorities. *Journal of Applied Ecology*, 46: 888-896.
7	Armenteras, D., Rodriguez, N., Retana, J. 2015. National and regional relationships of carbon storage and tropical biodiversity. *Biological Conservation*, 192: 378-386.
8	Bai, Y., Ouyang, Z.Y., Zheng, H., Li, X.M., Zhuang, C.W., Jiang, B. 2012. Modeling soil conservation, water conservation and their tradeoffs: A case study in Beijing. *Journal of Environmental Sciences-China,* 24: 419-426.
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10	Balbi, S., del Prado, A., Gallejones, P., Geevan, C.P., Pardo, G., Perez-Minana, E., Manrique, R., Hernandez-Santiago, C., Villa, F. 2015. Modeling trade-offs among ecosystem services in agricultural production systems. *Environmental Modelling & Software,* 72: 314-326.
11	Baral, H., Keenan, R.J., Fox, J.C., Stork, N.E., Kasel, S. 2013. Spatial assessment of ecosystem goods and services in complex production landscapes: A case study from south-eastern Australia. *Ecological Complexity*, 13: 35-45.
12	Baraloto, C., Alverga, P., Quispe, S.B., Barnes, G., Chura, N.B., da Silva, I.B., Castro, W., da Souza, H., Moll, I.D., Chilo, J.D., Linares, H.D., Quispe, J.G., Kenji, D., Medeiros, H., Murphy, S., Rockwell, C.A., Shenkin, A., Silveira, M., Southworth, J., Vasquez, G., Perz, S. 2014. Trade-offs among forest value components in community forests of southwestern Amazonia. *Ecology and Society*, 19.
13	Bartomeus, I., Gagic, V., Bommarco, R. 2015. Pollinators, pests and soil properties interactively shape oilseed rape yield. *Basic and Applied Ecology*, 16: 737-745.
14	Biber, P., Borges, J.G., Moshammer, R., Barreiro, S., Botequim, B., Brodrechtova, Y., Brukas, V., Chirici, G., Cordero-Debets, R., Corrigan, E., Eriksson, L.O., Favero, M., Galev, E., Garcia-Gonzalo, J., Hengeveld, G., Kavaliauskas, M., Marchetti, M., Marques, S., Mozgeris, G., Navratil, R., Nieuwenhuis, M., Orazio, C., Paligorov, I., Pettenella, D., Sedmak, R., Smrecek, R., Stanislovaitis, A., Tome, M., Trubins, R., Tucek, J., Vizzarri, M., Wallin, I., Pretzsch, H., Sallnas, O. 2015. How Sensitive Are Ecosystem Services in European Forest Landscapes to Silvicultural Treatment? *Forests,* 6: 1666-1695.
15	Boreux, V., Kushalappa, C.G., Vaast, P., Ghazoul, J. 2013. Interactive effects among ecosystem services and management practices on crop production: Pollination in coffee agroforestry systems. *Proceedings of the National Academy of Sciences of the United States of America,* 110: 8387-8392.

16	Brandt, P., Abson, D.J., DellaSala, D.A., Feller, R., von Wehrden, H. 2014. Multifunctionality and biodiversity: Ecosystem services in temperate rainforests of the Pacific Northwest, USA. *Biological Conservation,* 169: 362-371.
17	Briner, S., Huber, R., Bebi, P., Elkin, C., Schmatz, D.R., Gret-Regamey, A. 2013. Trade-Offs between Ecosystem Services in a Mountain Region. *Ecology and Society,* 18.
18	Brittain, C., Williams, N., Kremen, C., Klein, A.M. 2013. Synergistic effects of non-Apis bees and honey bees for pollination services. *Proceedings of the Royal Society B-Biological Sciences,* 280.
19	Butler, J.R.A., Wong, G.Y., Metcalfe, D.J., Honzak, M., Pert, P.L., Rao, N., van Grieken, M.E., Lawson, T., Bruce, C., Kroon, F.J., Brodie, J.E. 2013. An analysis of trade-offs between multiple ecosystem services and stakeholders linked to land use and water quality management in the Great Barrier Reef, Australia. *Agriculture Ecosystems & Environment,* 180: 176-191.
20	Cademus, R., Escobedo, F.J., McLaughlin, D., Abd-Elrahman, A. 2014. Analyzing Trade-Offs, Synergies, and Drivers among Timber Production, Carbon Sequestration, and Water Yield in Pinus elliotii Forests in Southeastern USA. *Forests,* 5: 1409-1431.
21	Carswell, F.E., Mason, N.W.H., Overton, J.M., Price, R., Burrows, L.E., Allen, R.B. 2015. Restricting new forests to conservation lands severely constrains carbon and biodiversity gains in New Zealand. *Biological Conservation,* 181: 206-218.
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23	Chung, M.G., Kang, H., Choi, S.U. 2015. Assessment of Coastal Ecosystem Services for Conservation Strategies in South Korea. *PLOS ONE,* 10.
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43	Frank, S., Furst, C., Pietzsch, F. 2015. Cross-Sectoral Resource Management: How Forest Management Alternatives Affect the Provision of Biomass and Other Ecosystem Services. *Forests,* 6: 533-560.
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56	Holt, A.R., Mears, M., Maltby, L., Warren, P. 2015. Understanding spatial patterns in the production of multiple urban ecosystem services. *Ecosystem Services*, 16: 33-46.
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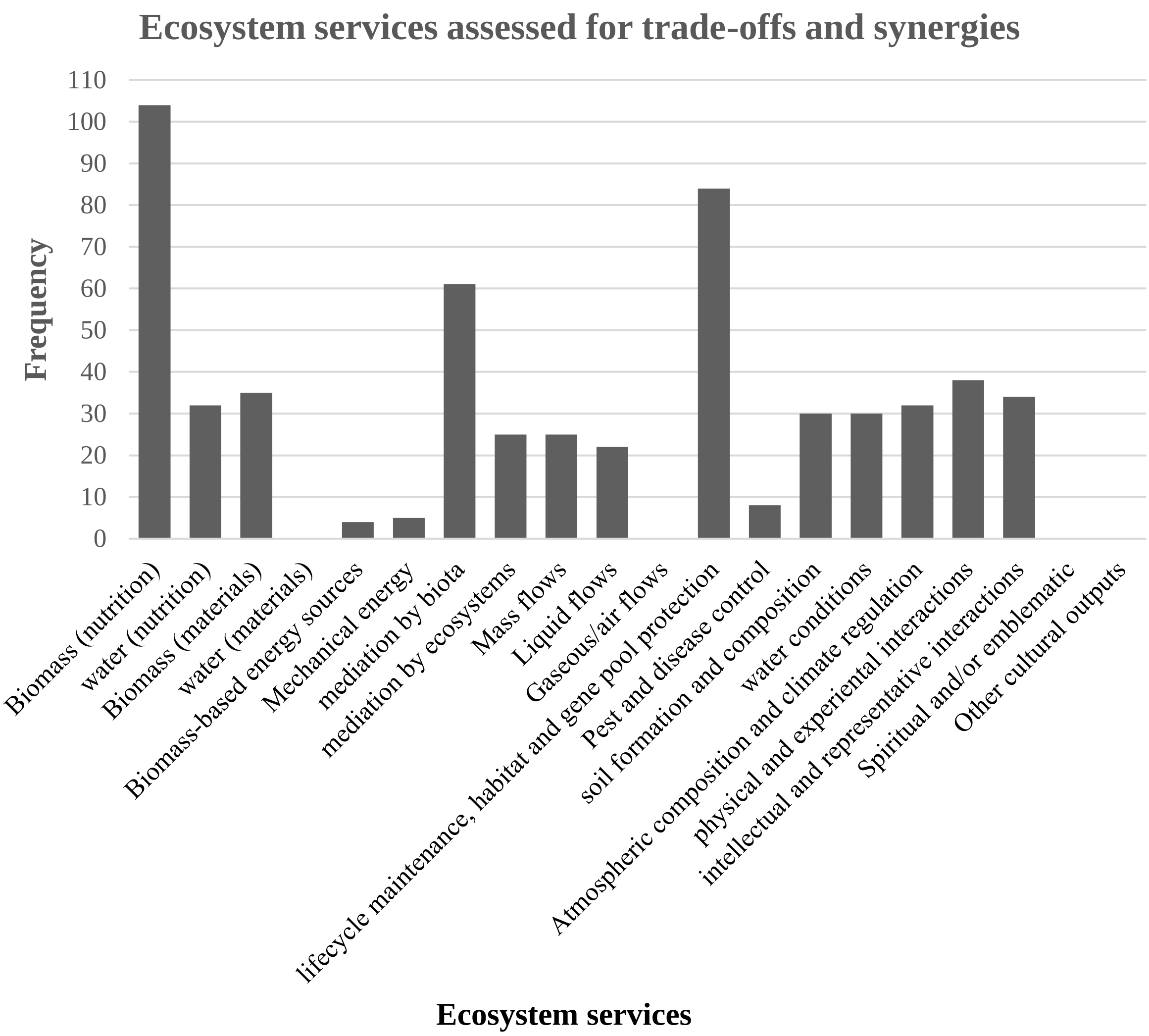


Figure C.1 Recorded number of ecosystem services assessed for trade-offs and synergies in the literature review database. Ecosystem service groups as defined by CICES classification V4.3 (http://cices.eu/).

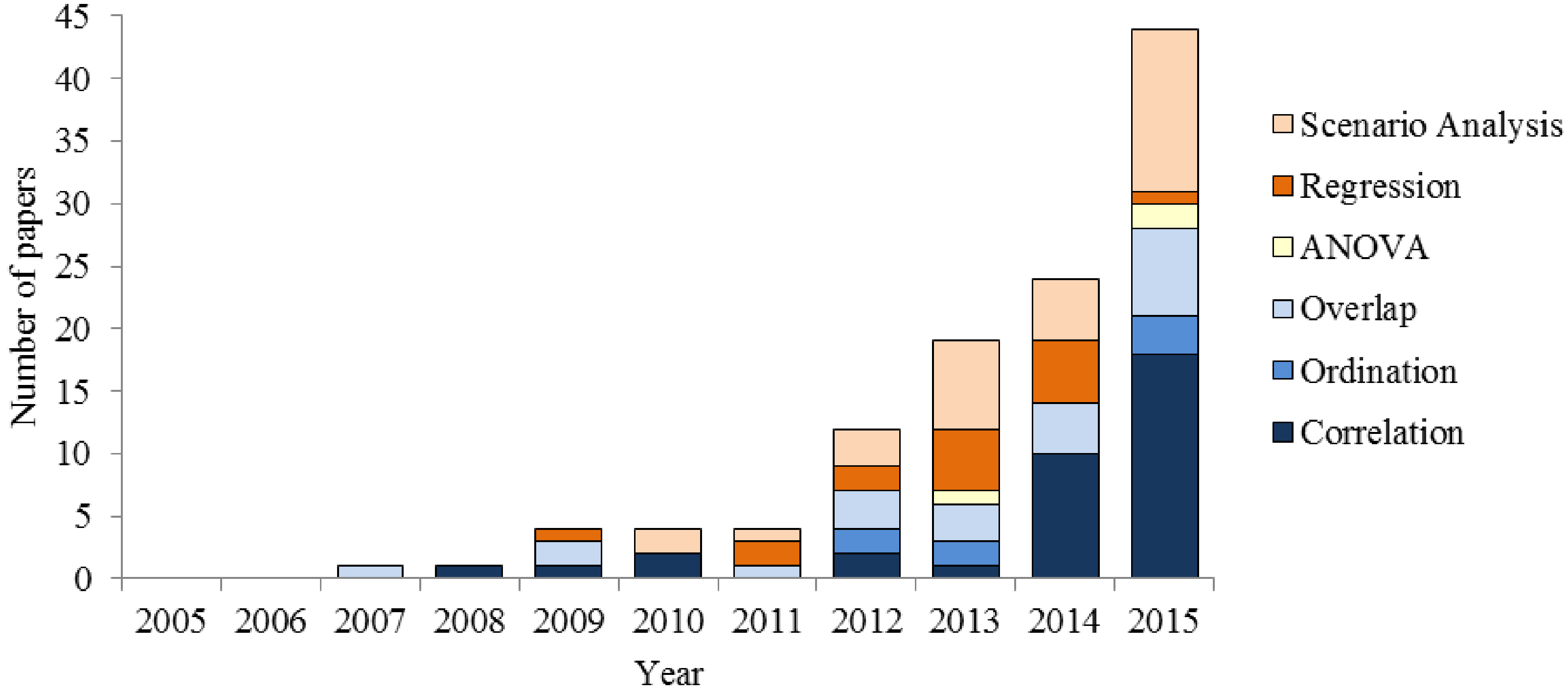


Figure C.2 Frequency of papers utilising each method to identify ecosystem service trade-offs and synergies between 2005 and 2015. ANOVA = analysis of variance.

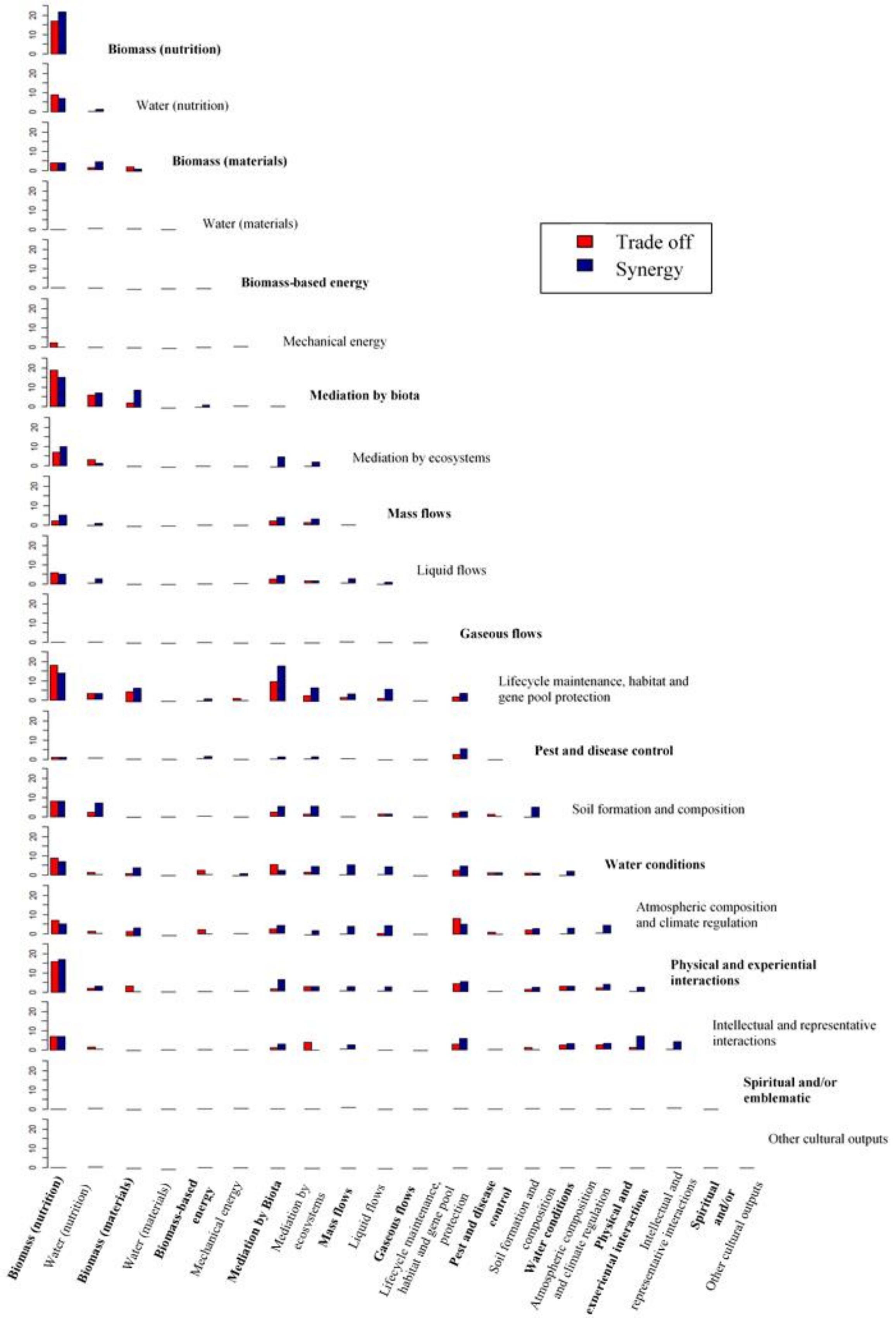


Figure C.3 The number trade-offs and synergies identified between ecosystem services in the literature database. Ecosystem service groups as defined by CICES classification V4.3 (http://cices.eu/).

APPENDIX D: SUPPLEMENTARY MATERIAL FOR CHAPTER 3

Table D.1 List of spatial datasets combined to create a dataset of parks within the Brisbane Local Governmental Area.

Dataset	Land use categories included as parks	Reference
Queensland Cadastre Dataset	Reserve; State Land	Department of Natural Resources and Mines (2017)[1]
Queensland Protected Areas	National Park; Conservation Park	Department of National Parks, Sport and Racing (2017)[2]
Queensland Recreation Areas	Gardens; Golf Course; Miscellaneous Area; Oval Area; Race Course; Race Track; Recreation Area; Show Ground; Zoo	Department of Natural Resources and Mines (2016)[1]
Brisbane City Council Park Classification Guide	Community Use Park; Corridor Link Park; Informal Use Park; Landscape Amenity Park; Natural Area Park; Other Park Land; Sport Park; Unclassified	Brisbane City Council (2006)[3]

[1] Department of Natural Resources and Mines. 2017. *Cadastral Data – Queensland series*. Government of Queensland, Brisbane, Australia. Online: https://data.qld.gov.au/dataset/cadastral-data-queensland-series

[2] Department of National Parks, Sports and Racing. 2017. *Protected Areas of Queensland*. Government of Queensland, Brisbane, Australia. Online: https://data.qld.gov.au/dataset/protected-areas-of-queensland-series

[3] Brisbane City Council. 2006. *Park Classification System Guide*, Brisbane City Council, Brisbane, Queensland, Australia.

<u>The survey questionnaire used to collect information on park visitation within Brisbane's parks</u>

Pre-mapping questions

(Appears after consent page, and prior to the mapping exercise)

Thanks! Before mapping, please answer the questions below that will help us customize the map for you.

1. **How did you learn about this study? (Please check one response.)**

☐ I received a request in the mail.

☐ I heard about the study from a relative, friend, or acquaintance.

☐ I read about the study in the Brisbane City Council newsletter.

☐ I heard about the study from a local community organisation.

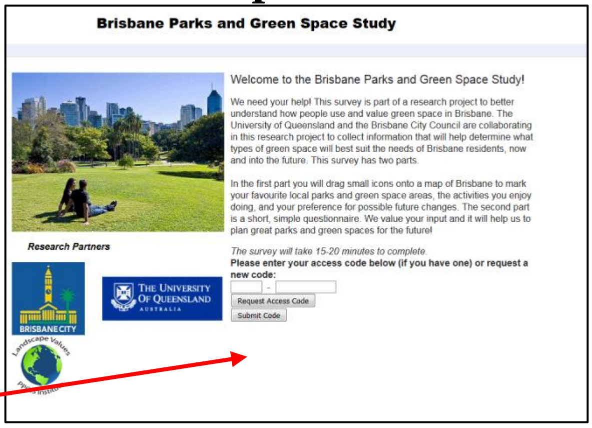
☐ Other (please write how you learned about the study) ____________________________

2. **Please enter the 4 digit post code where you live.**

Post code ☐

Click "continue" to begin the mapping exercise

Part 1
Mapping exercise for Green space in Brisbane



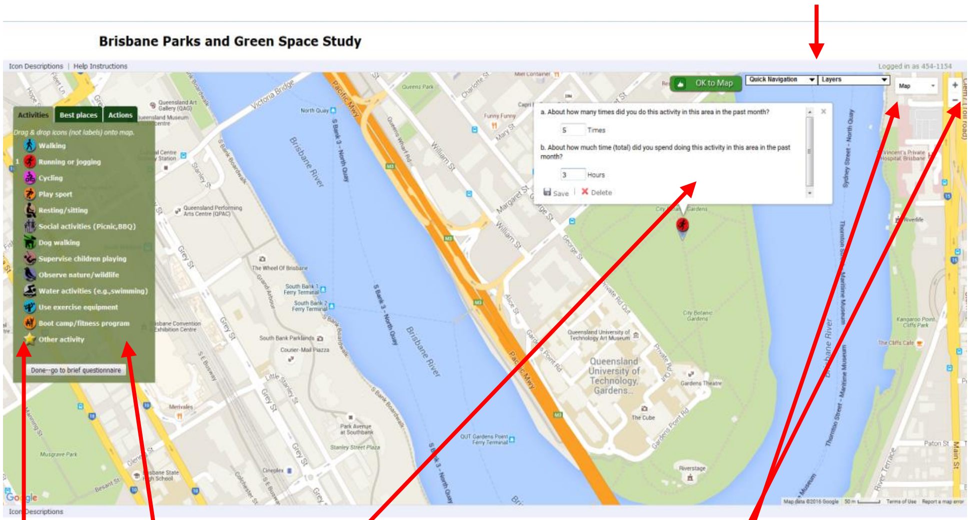
Instructions

1. Go to the website: http://www.landscapemap2.org/brisbaneparks

2. Enter the 7 digit access number provided in the letter of invitation (for random participants only)

Layers: You can view additional information about the study area

3. Drag and drop the markers onto the map showing their location on in the Brisbane study area and record time spent on the activity and the reason for doing this activity in the box that appears. Identify as many recreational activities as you can for the past 2 weeks. When satisfied, hit the "Done" button. ***Hint:*** you can delete or reposition markers.



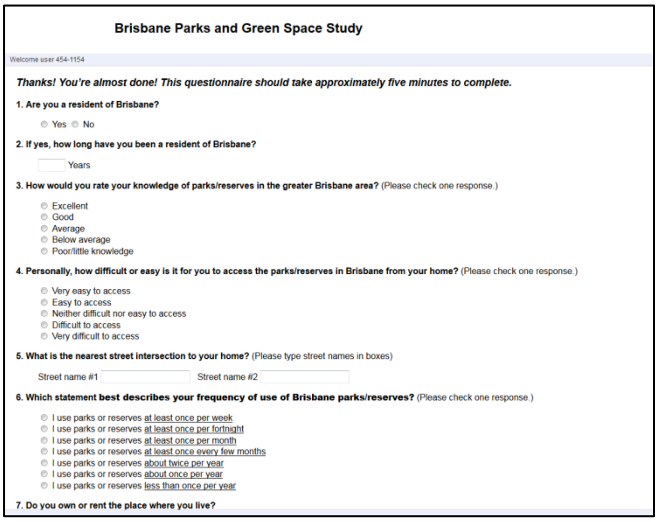
Drag these markers onto the Brisbane map

Click here to view marker descriptions

You can annotate each marker by mouse clicking on the marker

HINT: Google Maps allows you to zoom in and out of the region with 4 different map view options: Map, Terrain, Satellite, or Hybrid.

4. Answer the survey questions.



5. ***Thank you!*** If you provided an email address on the survey page, we will send you the study results. If you have any questions about the study, please contact: Associate Professor Jonathan Rhodes (j.rhodes@uq.edu.au).

PART 2

(This shows all combined questions asked for both RANDOM and VOLUNTEER participants. To view questions accessed only by RANDOM participants (participants who enter a unique access code to access the survey), please go to: http://www.landscapemap2.org/brisbaneparks/survey_v3.php. **To view questions accessed only by VOLUNTEER participants, please go to:**
http://www.landscapemap2.org/brisbaneparks/survey_v3vol.php)

Please tell us about yourself

The following questions are intended to tell us a little about the people who participate in this study and how they use their own yards or gardens for recreation. The information will only be used to compare the responses of different people, and how uses of their private green space differ, and you will not be individually identified in any way. However, if there is a question you do not want to answer for whatever reason, just leave it blank.

Thanks! You're almost done! This questionnaire should take approximately five minutes to complete.

1. **Are you a resident of Brisbane?**

   ☐ Yes ☐ No

2. **If yes, how long have you been a resident of Brisbane?**

   [ ] Years

3. **How would you rate your knowledge of parks/reserves and other green space in the greater Brisbane area?** (*Please check one response*)

   ☐ Excellent
   ☐ Good
   ☐ Average
   ☐ Below average
   ☐ Poor/little knowledge

4. **Personally, how difficult or easy is it for you to access parks/reserves or other green space in Brisbane from your home?** *(Please check one response)*
   - ☐ Very easy to access
   - ☐ Easy to access
   - ☐ Neither difficult nor easy to access
   - ☐ Difficult to access
   - ☐ Very difficult to access

5. **What is your street address, or if you prefer not to indicate your street address, the nearest street intersection to your home?** *(This information is <u>only</u> used to calculate distances to green spaces)*

Street address ☐

OR nearest intersection (two streets)

Street name #1 ☐ Street name #2 ☐

6. **Which statement best describes your frequency of use of parks/reserves and other green spaces?** *(Please check one response)*
   - ☐ I use parks/reserves/green spaces <u>at least once per week</u>
   - ☐ I use parks/reserves/green spaces <u>at least once per fortnight</u>
   - ☐ I use parks/reserves/green spaces <u>at least once per month</u>
   - ☐ I use parks/reserves/green spaces <u>at least once every few months</u>
   - ☐ I use parks/reserves/green spaces <u>about twice per year</u>
   - ☐ I use parks/reserves/green spaces <u>about once per year</u>
   - ☐ I use parks/reserves/green spaces <u>less than once per year</u>

7. **Do you own or rent the place where you live?**

   ☐ Own ☐ Rent

8. **Does your home/residence have a yard or garden?**

   ☐ Yes ☐ No → if “no”, skip to Question 10

9. **Think about the activities you do (or don't do) in your yard/garden. For each activity, indicate the number of times you have done the activity over the past two weeks.** *(There is no need to enter any responses for activities you don't do).*

Activity	How many times have you done this activity over the past two weeks?
Resting/sitting/relaxing	☐ Times
Social activities (e.g. BBQ)	☐ Times
Supervise children play	☐ Times
Exercise	☐ Times
Gardening or yard work	☐ Times
Observe nature/wildlife	☐ Times
Water activity (swimming)	☐ Times
Other activity (Please describe): ☐	☐ Times

10. **What is your sex/gender?**

☐ Male ☐ Female

11. **In what year were you born?** *(Please select one response)*

☐ Select a year from the drop down box

12. **Which of the following best describes the highest level of formal education you have completed?** *(Please select one response)*

☐ Select from drop down box *(options are: Postgraduate degree; graduate degree/Graduate Certificate; Bachelor Degree; Advanced Diploma/ Diploma; Certificate I – IV; Senior Secondary Education (yr 11 – 12); Junior Secondary Education)*

13. **Before tax, what is the *total* of all wages/salaries, government benefits, pensions, allowances and other income you *usually* receive?** *(Please select one response)*

☐ Select from drop down box *(options are: prefer not to say; $2000 or more a week; $1,500 - $1,999 a week; $1,250 - $1,499 a week; $1,000 - $1,249 a week; $800 – $999 a week; $600 - $799 a week; $400 - $599 a* *week; $300 - $399 a week; $200 - $299 a week; $1 - $199 a week; nil or negative income)*

14. Which life cycle category best describes you? *(Please check one response)*

[ ] Select from drop down box *(options are: Young single; mature single; Young couple with no children; mature couple with no children; Young family with youngest child less than 6 years old; middle family with youngest child 6-15 years old; senior family with youngest child over 16 years old; older couple with no children living at home; Grandparents responsible for grandchildren)*

15. What is your ancestry? *(Please check no more than 2 boxes)*

- ☐ English
- ☐ Irish
- ☐ Scottish
- ☐ Italian
- ☐ German
- ☐ Greek
- ☐ Chinese
- ☐ Australian
- ☐ Aboriginal or Torres Strait Islander
- ☐ Other -> Please specify
- ☐ Other -> Please specify

16. If you would like to receive a copy of the study results, please provide your email address below.

Email address [ ]

This completes the questionnaire! Thank you so much for your participation. As a small token of our appreciation for your time in completing this survey, you may select one of the options below to receive a $10 Coles/Myer gift voucher, or to have $10 donated to the charity of your choice below.

Note: For all gifts or donations, we do ask for an email to ensure we are only providing one gift or donation per valid email address. We take your privacy very seriously. We will separate any contact details provided from your responses and delete this information as soon as we have completed sending the gift or donation. We will provide a bulk payment to each of the charities at the completion of the survey.

☐ **$10 Coles/Myer Gift Card.** This voucher will be emailed to you (emailing date is xxxx following close of study). Coles/Myer Gift Cards are redeemable at a wide range of stores.

Please provide email address → ☐

☐ **$10 donation to WWF Australia.** WWF-Australia is part of the WWF International Network, the world's leading independent conservation organisation, which aims to stop the degradation of Australia's natural environment and to build a future in which humans live in harmony with nature. See http://www.wwf.org.au/

Please provide email address → ☐

☐ **$10 donation to Foodbank Queensland.** Foodbank is a non-denominational, charitable organisation which sources donated and surplus food from the food and grocery industry to distribute to welfare and community agencies that provide food assistance to people in need. See http://www.foodbankqld.org.au/

Please provide email address → ☐

☐ **$10 donation to the RSPCA.** RSPCA Qld is a non-government, community-based charity dedicated to protecting the welfare of all animals - great and small. Approximately 40,000 animals depend on us every year and we depend on your support and donations to continue our life-saving work. For more information see http://www.rspcaqld.org.au/

Please provide email address → ☐

Thank you again for your assistance in this survey. If you have any further comments, you can write them here (up to 250 words maximum).

☐

Click "submit" when finished

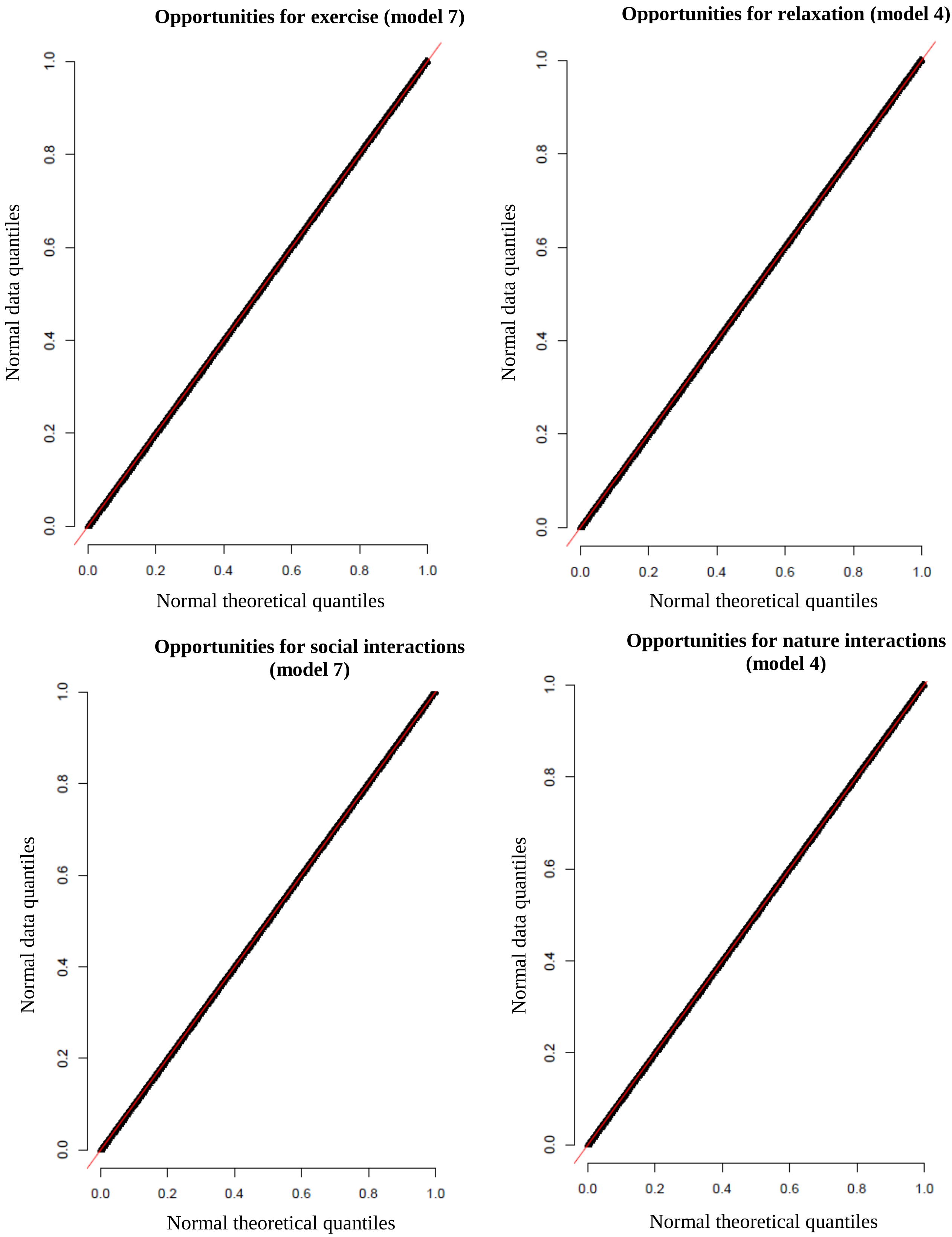


Figure D.1 Quantile-quantile plots for the parsimonious models for each ecosystem services. These plots were used as a measure of goodness of fit.

APPENDIX E: SUPPLEMENTARY MATERIAL FOR CHAPTERS 4 AND 5

Table E.1 Data sources for the predictor variables in each ecosystem service model.

Ecosystem service model	Predictor variable	Data source
Carbon storage	density of vegetation between 5-10m	LiDAR remotely sensed data (Mitchell et al. 2018[1])
	density of vegetation above 10m	LiDAR remotely sensed data (Mitchell et al. 2018[1])
	vertically dense canopy of high trees	LiDAR remotely sensed data (Mitchell et al. 2018[1])
	presence of mid-storey vegetation	LiDAR remotely sensed data (Mitchell et al. 2018[1])
Opportunities for exercise Opportunities for nature interactions	Distance from home (m)	Australian Bureau of Statistics census data (ABS 2016[2])
Opportunities for relaxation Opportunities for social interactions	Presence of amenities (Presence/absence of benches, shade devices, barbeques, toilets within park)	Park facility spatial dataset (Brisbane City Council 2015[3])
	Presence of play facilities (Presence/absence of child play equipment within park)	Park facility spatial dataset (Brisbane City Council 2015[3])
	Presence of access facilities (Presence/absence of paths within park)	Park facility spatial dataset (Brisbane City Council 2015[3])
	Exercise facilities (Presence/absence of exercise equipment within park)	Park facility spatial dataset (Brisbane City Council 2015[3])
	Presence of animal facilities (Presence/absence of off-leash dog zones within park)	Park facility spatial dataset (Brisbane City Council 2015[3])
	proportion of grass	LiDAR remotely sensed data (Chapter 3)
	Proportion of tree cover	LiDAR remotely sensed data (Chapter 3)
	Foliage height diversity	LiDAR remotely sensed data (Caynes et al. 2016[4])
	Area of park (m^2)	
	Shape of park (compactness)	Spatial data/FRAGSTATS (McGarigel et al. 2002[5])
Opportunities for exercise Opportunities for social interactions	Gender (male/female)	Australian Bureau of Statistics census data (ABS 2016[2])
	Education (highest level attained)	Australian Bureau of Statistics census data (ABS 2016[2])
	Weekly income (AUD)	Australian Bureau of Statistics census data (ABS 2016[2])
	Age	Australian Bureau of Statistics census data (ABS 2016[2])

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